BIOLOGICAL EVALUATION / BIOLOGICAL ASSESSMENT

FOR

TERRESTRIAL AND AQUATIC WILDLIFE

YUBA TRAILS ENHANCEMENT

YUBA RIVER RANGER DISTRICT
TAHOE NATIONAL FOREST

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PREPARED BY: MARILYN TIERNEY
DISTRICT WILDLIFE BIOLOGIST

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I. EXECUTIVE SUMMARY--March 15, 2018; Yuba Trails Enhancement Project

SCOPE OF AREA AFFECTED: The Project is located in Sierra County, in the Yuba River Ranger District, north of Downieville, CA, within multiple sections of Township 20 and 21N, Range 11E. Project would (1) reroute portions of 4 single track motorized trails by decommissioning and restoring 5 miles of steep trail and constructing 7 miles of lower-gradient trail.

Table 1. Executive summary of species considered for analysis in this Biological Evaluation/Biological Assessment and the effects determination.

SPECIES	SPECIES STATUS[1]	OCCURS OR HAS SUITABLE HABITAT WITHIN THE PROJECT AREA	EFFECTS DETERMINATION[2]
Western bumblebee	S	Yes	May affect, no trend toward listing
Bald eagle	S	Yes	No effect
California spotted owl	S	Yes	May affect, no trend toward listing
Great gray owl	S	No	No effect
Northern goshawk	S	Yes	May affect, no trend toward listing
Willow flycatcher	S	No	No effect
Greater sandhill crane	S	No	No effect
Pacific marten	S	Habitat	May affect, no trend toward listing
North American wolverine	S	Habitat	No effect
Pallid bat	S	Habitat	No effect
Townsend's big-eared bat	S	Habitat	May affect, no trend toward listing
Fringed myotis	S	Habitat	May affect, no trend toward listing
AQUATIC SPECIES			
California red-legged frog	T	No	No effect
Lahontan cutthroat trout	T	No	No effect
Western pond turtle	S	No	No effect
Foothill yellow-legged frog	S	Yes	No effect
Sierra Nevada yellow-legged frog	E	No	No effect
Great Basin rams-horn snail	S	No	No effect
Lahontan Lake tui chub	S	No	No effect
Hardhead	S	No habitat	No effect
California floater	S	No habitat	No effect
Black juga	S	No habitat	No effect

[1]Key: E = USFWS Endangered, T = USFWS Threatened, PE = USFWS Proposed Endangered (pending final rule), PT = USFWS Proposed Threatened (pending final rule), C = USFWS Candidate (warranted but precluded), S = USFS R5 Sensitive.
[2]Effects determinations for sensitive species of "may affect individuals, but is not likely to result in a trend toward Federal listing or loss of viability" is shown as NLRT; "may affect not likely to adversely affect" for proposed or Federally-listed threatened or endangered species is shown as NLAA.

II. INTRODUCTION

The purpose of this Biological Evaluation/Biological Assesment is to document analysis of the potential effects of the Yuba Trails Enhancement Project on Forest Service Sensitive Species, and U.S. Fish and Wildlife's designated threatened, endangerd, candidate, and proposed aquatic and terrestrial wildlife species and their habitats:

The above list includes United States Department of Interior Fish and Wildlife Service (USFWS) threatened, endangered, candidate, and proposed species maintained at 50 CFR 17.11 (Last updated September 5, 2017), and United States Department of Agriculture (USDA) Forest Service Region 5 Forester's Sensitive Species, listed for the Tahoe National Forest (Updated as of September 9, 2013). This evaluation does not address the following USFWS threatened or endangered species: *Coccyzus americanus* (yellow-billed cuckoo (T)), *Branchinecta lynchi* (Vernal Pool fairy shrimp (T)), *Lepidurus Packardt* (Vernal Pool tadpole shrimp (E)), *Chasmistes cujus* (cui-ui (E)), *Hypomesus transpacificus* (delta smelt (T)), or *Oncorhynchus mykiss* (steelhead (T)) because the Tahoe National Forest is outside of the range or does not contain habitat for these species. *Oncorhynchus mykiss* (Central Valley steelhead, Sacramento River) do not occur in the Tahoe National Forest due to dams. If a project in the Tahoe National Forest could potentially have effects to any of these species they would be analyzed in detail.

This biological evaluation was prepared in accordance with Forest Service Manual (FSM) direction 2672.42 and meets legal requirements set forth under Section 7 of the Endangered Species Act of 1973, as amended, and implementing regulations [19 U.S.C. 1536 (c), 50 CFR 402.12 (f) and 402.14 (c)].

Literature cited and references throughout this biological evaluation can be found at the end of the document, first by general resource documents then by individual species or species group.

III. CONSULTATION TO DATE

The Fish and Wildlife Service is contacted every 90 days to obtain a current list of threatened, endangered, proposed and candidate species that may be present in the Tahoe National Forest. The most recent list was reported September 5, 2017 and is available for review at the District Office.

Forest plans for national forests lying within the Sierra Nevada were amended under the Sierra Nevada Forest Plan Amendment (USDA Forest Service 2001 and 2004). The Regional Forester consulted with the California and Nevada Operations Office of Fish and Wildlife Service for that amendment. The Biological Opinion is dated January 11, 2001. The determination in the biological opinion is that the selected action is not likely to jeopardize the continued existence of species listed pursuant to the Act (bald eagle (subsequently delisted), California red-legged frog, and Lahontan cutthroat trout). No terms or conditions were provided. Conservation recommendations are discussed in the corresponding species portions of this Biological Evaluation/Biological Assessment if they are applicable to the Tahoe National Forest species and management activities.

IV. CURRENT MANAGEMENT DIRECTION

Current management direction on desired future conditions for Threatened, Endangered and Sensitive species in the Tahoe National Forest can be found in the following documents, filed at the District Office:

- Forest Service Manual and Handbooks (FSM/FSH 2670)
- National Forest Management Act (NFMA)
- Endangered Species Act (ESA)
- National Environmental Policy Act (NEPA)
- Tahoe National Forest Land and Resource Management Plan (1990), as amended by the 1999 Record of Decision for the Herger-Feinstein Quincy Library Group Forest Recovery Act Final Environmental Impact Statement (HFQLG) [as revised by the 2003 Record of Decision for the Herger-Feinstein Quincy Library Group Forest Recovery Act Final Supplemental Environmental Impact Statement], and the 2004 Record of Decision for the Sierra Nevada Forest Plan Amendment Final Supplemental Environmental Impact Statement.
- Species specific Recovery Plans which establish population goals for recovery of those species
- Species management plans
- Species management guides or conservation strategies
- Regional Forester policy and management direction

The Tahoe National Forest Land and Resource Management Plan (LRMP; 1990) was amended in 2001 by the Record of Decision for the Sierra Nevada Forest Plan Amendment (SNFPA 2001; USDA Forest Service 2001), which was then replaced in its entirety by the 2004 Record of Decision for the Sierra Nevada Forest Plan Amendment Final Supplemental Environmental Impact Statement (SNFPA 2004; USDA Forest Service 2004). Detailed information including specific standards and guidelines for species management can be found in the SNFPA 2004. General Forest Service direction for Threatened, Endangered, and Sensitive species is summarized below:

FSM 2670.31 THREATENED AND ENDANGERED SPECIES

1) Place top priority on conservation and recovery of endangered, threatened, and proposed species and their habitats through relevant National Forest System, State and Private Forestry, and Research activities and programs.
2) Establish through the Forest planning process objectives for habitat management and/or recovery of populations, in cooperation with States, the USFWS, and other Federal agencies.
3) Through the biological evaluation process, review actions and programs authorized, funded, or carried out by the Forest Service to determine their potential for effect on threatened and endangered species and species proposed for listing.
4) Avoid all adverse impacts on threatened and endangered species and their habitat except when it is possible to compensate adverse effect totally through alternatives identified in a biological opinion rendered by the USFWS, or when the USFWS biological opinion

recognizes an incidental taking. Avoid adverse impacts on species proposed for listing during the conference period and while their Federal status is being determined.

5) Initiate consultation or conference with the USFWS when the Forest Service determines that proposed activities may have an adverse effect on threatened, endangered, or proposed species or when Forest Service projects are for the specific benefit of a threatened or endangered species
6) Identify and prescribe measures to prevent adverse modification or destruction of critical habitat and other habitats essential for the conservation of endangered, threatened, and proposed species. Protect individual organisms or populations from harm or harassment as appropriate.

FSM 2670.32 SENSITIVE SPECIES

1) Assist States in achieving their goals for conservation of endemic species.
2) As part of the National Environmental Policy Act process, review programs and activities, through a biological evaluation, to determine their potential effect on sensitive species.
3) Avoid or minimize impacts to species whose viability has been identified as a concern.
4) If impacts cannot be avoided, analyze the significance of potential adverse effects on the population or its habitat within the area of concern and on the species as a whole.
5) Establish management objectives in cooperation with the States when a project on National Forest System lands may have a significant effect on sensitive species population numbers or distribution. Establish objectives for Federal candidate species, in cooperation with the USFWS and the States.

V. DESCRIPTION OF THE PROPOSED PROJECT AND ALTERNATIVES

BACKGROUND

The Forest Service is proposing the Yuba Trails Enhancement Project to: (1) re-route four motorized trail segments; (2) remove three existing unauthorized routes; and (3) construct two connector motorized trails on National Forest System lands on the Yuba River Ranger District of the Tahoe National Forest. These actions are needed to address ongoing soil erosion and potential water quality impacts associated with existing steep, motorized trails and unauthorized routes in this area. The proposed actions would address soil erosion and water quality concerns, provide for a sustainable trail system, improve motorized recreation opportunities, and enhance trail users' experiences.

The Project area is located north of the community of Downieville (T 20 & 21N, R10E & R11E, multiple sections) and entirely within Sierra County (Figure 1). The four proposed motorized trail re-routes and three unauthorized route removals are located within the East and West Yuba Inventoried Roadless Areas. The two proposed motorized connector trails lie outside the Inventoried Roadless Areas.

PURPOSE AND NEED

Over the past several years, motorized recreational use has been steadily increasing in the Project area. This use has created localized impacts for both recreation users and natural resources, including deteriorating conditions of system trails; adverse watershed and soil impacts; and concerns regarding recreational experiences.

The purposes of the Yuba Trails Enhancement Project are to: respond to resource impacts from increasing and changing demands for motorized trail use and other motorized dispersed recreation; ensure the managed trail system is sustainable; and enhance motorized recreation experiences. Implementation of the Project would result in enhanced recreation opportunities and experiences; provide additional outfitter and guiding opportunities; mitigate existing impacts to soil and water resources; and address community interests and public safety. The proposed Project is consistent with Forest Plan management direction in the *Tahoe National Forest Land and Resource Management Plan* (1990) as amended by the *Sierra Nevada Forest Plan Amendment Record of Decision* (SNFPA ROD 2004).

PROPOSED ACTION

The proposed action has three components: (1) four motorized trail re-routes followed by subsequent decommissioning and restoration of the replaced trail sections; (2) removal of three existing unauthorized routes through restoration to a natural state; and (3) construction of two connector motorized trails.

Trail Re-Routes

The four proposed motorized trail reroutes are designed to eliminate problems associated with overly steep and heavily eroding portions of the Rattlesnake/Downie River Trail, Pauley Creek Trail, Big Boulder Trail, and Lavezzola Trail as follows:.

Downie River / Rattlesnake Trail : The Project would re-route approximately 1.5-miles of the steep existing Rattlesnake trail segment (25-40% grades) that is intercepting a drainage, and replace it with an approximately 2.5 miles of multiple use motorized single track trail with a grade of 5-10%. With the proposed re-route, the entire trail length will be called the Downie River Trail. The Project would decommission and restore to natural grade the existing 1.5-mile steep section of trail (25-40% grades).

Pauley Creek Trail : The Project would re-route approximately 0.5 miles of a steep existing trail segment (25-35% grades) that climbs straight up and replace it with approximately 0.5 miles of multiple use motorized single track trail with grades of 5-10%. The Project would decommission and restore to natural grade 0.5 miles of a steep existing section of trail (25-35% grades).

Lavezzola Trail: The Project would re-route approximately 2 miles of a steep existing trail segment (25-40% grades) that climbs straight up and replace it with approximately 3 miles of multiple use motorized single track trail with grades of 5-10%. The Project would decommission and restore to natural grade the 2-mile steep existing section of trail (25-40% grades).

Big Boulder Trail: The Project would re-route approximately 1 mile of a steep existing trail segment (25-40% grades) that climbs straight up and replace it with approximately 1 mile of multiple use motorized single track trail with grades of 5-10%. The Project would decommission and restore to natural grade the 1-mile steep existing section of trail (25-40% grades).

Connector Trails

Second Divide Trail: This proposed 0.15-mile connector trail is designed to enhance the safety and experience of users on First, Second and Third Divide Trails by building a trail that bypasses the County Road that connects the two popular trails. Currently a legal connection for motorcycle users does not exist as green sticker motorcycles are not allowed on roads not classified for their use under the *Tahoe National Forest Motorized Travel Management Record of Decision* (September 2010).

This connector trail would also improve the user experience by creating a continuous single track trail that extends Second Divide Trail with a connection to First Divide Trail. The new multiple use motorized single track trail would be designed with grades of 5-10%.

Unauthorized Route Restoration

The unauthorized route to Sisson Mine, an unauthorized route near Hawley Meadow (old Gold Valley), and unauthorized route near Butcher Ranch would be removed and the land restored to a natural grade. These routes are not needed for public use and are unsustainable. Approximately 3 miles of unauthorized routes would be restored to natural conditions.

Trail Construction Standards

General

Trail work will occur through the use of hand work or by qualified machine operators approved by USFS. Any trail work other than standard maintenance will be approved by the recreation officer prior to commencement.

Standards

- Average Grade Pitch: 5% (within aprox. 100ft or overall segment) grade reversal every 100-200ft
- Moderate duration pitches (50ft): 15% max, include grade reversal or out-slope feature
- Short steep slopes (25ft): 25% max, include grade reversal or out-slope feature
- The intent on pitch limiters is to create sustainable trail, volume of usage, soil or surface type; hydrology and user types may affect design standards. Steeper segments may be approved with hardened bench elements.
- Bench Width: 24” - 36”
- Clearance from trail center: 30” for general obstructions
- Brush removal from trail center: 5ft
- Height clearance: 7ft

Guidelines for preventing Resource Damage

- Build on side slopes
- Avoid ridge-top or fall line alignments
- Stay out of meadows or flatlands where drainage is poor
- Favor the upslope of trees to prevent root damage
- Build mild, undulating trail alignment that utilizes frequent grade reversals
- Out-slope bench when possible
- Camber outside of turns to minimize lateral wear
- Avoid over-pitch alignments
- Create good sight lines
- Design intuitive trail alignments

Creek or ephemeral drainage crossings

- Locate crossings at stable locations
- Trail at crossing should always be at least 12+ inches lower than approach from either side
- Harden active crossings with cobbled rock to minimize creek disturbance
- If a bridge is used, construct so freeboard is above 100 year mark

- If bridge footings are within 100 year mark, embed into embankment 2ft or more to avoid high water scouring

Switchbacks and Rolling Turns

- Provide grade reversals within 50ft of both sides of turn and stage so that lower grade reversals catch upper drainage runoff.
- Rolling turns have radius's in excess of 4ft tc (trail center) and occur on slopes which are less than 30%
- Switchbacks have radius's of less than 4ft tc and occur on slopes greater than 30%
- Anticipate approaches to turns and design speed reduction to eliminate skid bumps
- Keep overall switchback radius bench at 5-10% max to minimize wear
- If cambering turn, leave flat climbing radius towards center
- Locate turn in spot that limits short cutting
- Separate trails from each other as early as possible

Rolling dips, Grade Reversals or Drain Dips

- Downhill rise should be 6-12" above low point
- Features should be 10-20 ft in length for smooth transitions
- Place at all ephemeral (rarely active) or seasonal drainages

Bermed Turns

- Confirm all turns drain by splitting or tilting the turn on the slope
- Leave un-cambered inside space for hiking or uphill riding

Evaluate safety and confirm berm is free of encroaching hazards like trees or rocks

VI. EXISTING ENVIRONMENT, EFFECTS OF THE PROPOSED ACTION AND ALTERNATIVES, AND DETERMINATION

SPECIES-SPECIFIC ANALYSIS AND DETERMINATION

This section discusses each species in three parts, A) Existing Environment, B) Effects of the Proposed Action and Alternatives including Project Design Standards, and C) Conclusion and Determination.

Section A describes the existing environment including species life history, status, and relevant information. Further detail can be found in the Sierra Nevada Forest Plan Amendment Final Environmental Impact Statement and Record of Decision (SNFPA 2001; USDA Forest Service 2001) and Sierra Nevada Forest Plan Amendment Record of Decision and Final Supplemental Environmental Impact Statement and Record of Decision (SNFPA 2004; USDA Forest Service 2004).

Section B addresses the effects of the proposed project to the various species including project design standards and required mitigation measures. Effects are described as direct, indirect or cumulative. Direct effects as described in this evaluation refer to mortality or disturbance that results in flushing, displacement or harassment of the animal. Indirect effects refer to modification of habitat and/or effects to prey species. Cumulative effects represent "The impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions" (National Environmental Policy Act 1986).

If the cumulative effects involve a federally listed species, the definition of cumulative effects expands to address "those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation" (Endangered Species Act, 1973 as amended).

Regulations at 50 CFR 402.02 in regards to federally listed species are as follows:

> Effects of the action refers to the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration.

For NEPA, "Connected action" as defined in CEQ Section 1508.25(a):

> Actions are connected if they: (1) automatically trigger other actions which may require environmental impact statements; (2) cannot or will not proceed unless other actions are taken previously or simultaneously, or; (3) are interdependent parts of a larger action and depend on the larger action for their justification.

Section C provides a summary of supporting conclusions and the statement of determination for each species based upon relevant information provided in Sections A & B.

<u>Terrestrial Species</u>

WESTERN BUMBLE BEE

Status: USFS R5 Sensitive

A. Western Bumble Bee: Existing Environment

The western bumble bee (*Bombus occidentalis)* is experiencing severe declines in distribution and abundance due to a variety of factors including diseases and loss of genetic diversity

(Tommasi et al. 2004, Cameron et al. 2011, Koch et al. 2012). Historically, the species was broadly distributed across western North America along the Pacific Coast and westward from Alaska to the Colorado Rocky Mountains (Thorp and Shepard 2005, Koch et al. 2012) and was one of the most broadly distributed bumble bee species in North America (Cameron et al. 2011). The current range includes California and adjacent states. Six bumble bee occurrences are documented on the Tahoe National Forest prior to 2000 (www.xerces.org).

Bumble bees introduced from Europe for commercial pollination apparently carried a microsporidian parasite, *Nosema bombi*, which have spread to native bumble bee populations. Highest incidences of declining *B. occidentalis* populations are associated with highest infection rates with the *Nosema* parasite, and the incidence of *Nosema* infestation is significantly higher in the vicinity of greenhouses that use imported bumble bees for pollination of commercial crops (Cameron et al. 2011).

Although the general distribution trend is steeply downward, especially in the west coast states, some isolated populations in Oregon and the Rocky Mountains appear stable (Rao et al. 2011, Koch et al. 2012). The overall status of populations in the west is largely dependent on geographic region: populations west of the Cascade and Sierra Nevada mountains are experiencing steeply declining numbers, while those to the east of this dividing line are more secure with relatively unchanged population sizes. The reasons for these differences are not known.

Bumble bees are threatened by many kinds of habitat alterations that may fragment or reduce the availability of flowers that produce the nectar and pollen they require, and decrease the number of abandoned rodent burrows that provide nest and hibernation sites for queens. Major threats that alter landscapes and habitat required by bumble bees include agricultural and urban development. Exposure to organophosphate, carbamate, pyrethroid and particularly neonicotinoid insecticides has recently been identified as a major contributor to the decline of many pollinating bees, including honey bees and bumble bees (Henry et al. 2012, Hopwood et al. 2012). In the absence of fire, native conifers encroach upon meadows, which also decrease foraging and nesting habitat available for bumble bees.

According to studies done in England (Goulson et al. 2008), grazing during the autumn and winter months may provide excellent bumble bee habitat and prevent the accumulation of coarse grasses. Heavy grazing and high forage utilization can negatively impact bumble bees since flowering plants providing necessary nectar and pollen may become unavailable, particularly during the spring and summer when queens, workers and males are all present and active.

Western bumble bee queens overwinter in the ground in abandoned rodent (i.e. mouse, chipmunk or vole) nests at depths from 6-18 inches and typically emerge about mid-March. The queen then lays fertilized eggs and nurtures a new generation. She first creates a thimble-sized and shaped wax honey pot, which she provisions with nectar-moistened pollen for 8-10 individual first-generation workers when they hatch. The larvae will receive all of the proteins, fats, vitamins and minerals necessary for growth and normal development from pollen. Eventually all the larvae will spin a silk cocoon and pupate in the honey pot. The workers that emerge will begin foraging and provisioning new honey pots as they are created to accommodate additional recruits to the colony. Individuals emerging from fertilized eggs will become workers that reach peak abundance during July and August. Foraging individuals are largely absent by the end of September. Those that emerge from unfertilized eggs become males, which do not forage and only serve the function of reproducing with newly emerged queens. During the season, a range of 50 to hundreds of individuals may be produced depending on the quantity and quality of flowers available. When the colony no longer produces workers, the old queen will eventually die and newly emerged queens will mate with males and then disperse to found new colonies. During this extended flight that may last for up to two weeks she may make several stops to examine the ground for a suitable burrow. Mikkola (1984) reported that bumble bees may forage up to a distance of 80 kilometers (50 miles) in Finland (Heinrich 1979). Where nesting habitat is scarce, bumble bee species having queens that emerge early (mid-March) in the season like *B. vosnesenskii* which co-occurs with the later emerging *B. occidentalis*, may be able to monopolize available nest sites and reduce the chances of success for bumble bee species emerging later.

Western bumble bees have a short proboscis or tongue length relative to other co-occurring bumble bee species, which restricts nectar gathering to flowers with short corolla lengths and limits the variety of flower species it is able to exploit. Western bumble bees have been observed taking nectar from a variety of flowering plants, including *Aster* spp., *Brassica* spp., *Centaurea* spp., *Cimicifuga arizonica*, *Corydalis caseana*, *Chrysothamnus* spp., *Cirsium* spp., *Cosmos* spp., *Dahlia* spp., *Delphinium nuttallianum*, *Erica carnea*, *Erythronium grandiflorum*, *Foeniculum* spp., *Gaultheria shallon*, *Geranium* spp., *Gladiolus* spp., *Grindelia* spp., *Haplopappus* spp., *Hedysarum alpinum*, *Hypochoeris* spp., *Ipomopsis aggregata*, *Lathyrus* spp., *Linaria vulgaris*, *Lotus* spp., *Lupinus monticola*, *Mentha* spp., *Medicago* spp., *Melilotus* spp., *Mertensia ciliata*, *Monardella* spp., *Nama* spp., *Origanum* spp., *Orthocarpus* spp., *Pedicularis capitata*, *P. kanei*, and *P. langsdorfii*, *P. groenlandica*, *Penstemon procerus*, *Phacelia* spp., *Prunus* spp., *Raphanus* spp., *Rhododendron* spp., *Salix* spp., *Salvia* spp., *Solidago* spp., *Symphoricarpos* spp., *Tanacetum* spp., *Taraxacum* spp., *Trifolium dasyphyllum*, *Trichostema* spp., *Trifolium* spp. and *Zea* spp. (Evans et al. 2008).

In a study (Helen et. al 2017) conducted by the Institute for Bird Populations in the central Sierra Nevada (Eldorado National Forest) that anyalyzed the use of post-fire montane chapparel by multiple bumble bee species, they concluded that openings containing herbaceouse vegetation and riparian areas had a greater bee abundance versus areas of upland chaparral vegetation. Of the chapparel species, bumble bees foraged on bearclover and Phacelia significantly more than expected based on their availability, and on whitethorn, manzanita, and deerbrush species significantly less then expected based on availability.

Unlike all other bees, bumble bees are large enough to be capable of thermoregulation, which allow them to maintain their foraging activities for longer periods of the day, but also to occupy regions with more extreme latitudes and temperatures compared to other bees (Heinrich 1979). Bumble bees may continue to forage when temperatures are below freezing even in inclement weather (Heinrich 1979).

B. Western bumble bee: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct and Indirect

Constructing a new trail could remove flowering plants upon which bumblebees forage. Individual bees could also be displaced from holes and woody structures that they utilize for hiding and sheltering. Indirect effects that remove hiding and sheltering habitat could also occur during new trail construction and relocation. The majority of new and relocated trail traverses mixed conifer forests of varying canopy closures and shrub habitats that provide less bumblebee habitat than meadows where flowering plants are more abundant. This project would not occur within any meadow habitat. Therefore, although direct and indirect impacts could occur, they occur outside of preferred habitat for this species.

Cumulative

The spatial area selected for analysis is where trails would be relocated, new trails placed, and decommissioned trails, buffered by 0.25 miles. Cumulative effects to this species within this project area could occur from road and trail maintenance, dispersed recreational uses off of trails, and vegetation management. The majority of the project area occurs within the East-West Yuba RARE II area, within which vegetation management is focused around reducing vegetation encroaching onto roads and trails and felling hazard trees. There are no planned vegetation management projects within the project area that would add cumulative effects. In general, few cumulative effects that impact individual bumblebees or their habitats occur. This project may add a small amount of cumulative effects, but because the trail is not located within high quality habitat, the degree of these effects is small.

C. Western bumble bee Conclusion and Determination

It is my determination that this project may affect, but will not lead to a trend toward listing for the western bumblebee.

BALD EAGLE

Status: USFS R5 Sensitive

A. Bald Eagle: Existing Environment

Pertinent regulatory history and status:

- 2007: Bald eagle delisted from the List of Endangered and Threatened Wildlife (USFWS 2007d; 72 FR 37346). At the time of delisting, the bald eagle was placed on the USFS R5 Sensitive Species List. In anticipation of delisting the bald eagle, the U. S. Fish and Wildlife Service issued National Bald Eagle Management Guidelines (USFWS 2007a), a regulatory definition of "disturb" under the Bald and Golden Eagle Protection Act (USFWS 2007b; 72

FR 31132), and proposed new permit regulations to authorize take under the Bald and Golden Eagle Protection Act (USFWS 2007c; 72 FR 31141).

- Bald eagles continue to be protected under the Migratory Bird Treaty Act and the Bald and Golden Eagle Protection Act (Eagle Act).

Bald eagle nesting and wintering habitat occurs throughout the Pacific Southwest Region, which includes both the Sierra Nevada and Klamath Provinces. The Tahoe National Forest LRMP (Forest Plan) outlines management of bald eagle nesting and wintering habitats for target populations as specified in the species recovery plan. A Tahoe National Forest Bald Eagle Management Plan (April 2, 2004) was submitted to the USFWS. The SNFPA provided no new standards and guidelines for bald eagle management. Conservation recommendations from the Biological Opinion for the SNFPA (FWS 2001) are included as management recommendations within the Tahoe National Forest Bald Eagle Management Plan. The USFWS published National Bald Eagle Management Guidelines in May 2007.

Nesting territories are normally associated with lakes, reservoirs, rivers or large streams (Lehman 1979). Bald eagle nests are usually located in uneven-aged (multi-storied) stands with old growth components (Anthony et al. 1982). Most nests in California are located in predominantly coniferous stands. Factors such as relative tree height, diameter, species, and position on the surrounding topography, distance from water, and distance from disturbance also appear to influence nest site selection (Grubb 1976, Lehman et al. 1980, Anthony and Isaacs 1989).

Trees selected for nesting are characteristically one of the largest in the stand or at least codominant with the overstory. Nest trees usually provide an unobstructed view of the associated water body and are often prominently located on the topography. Live, mature trees with deformed tops are occasionally selected for nesting. Of nest trees identified in California, about 71 percent were ponderosa pine, 16 percent were sugar pine, and 5 percent were incense cedar. The remaining 8 percent were distributed among five other coniferous species. Eagle nests may be located in snags, but most nests are probably constructed when trees were alive (Anthony and Isaacs 1989). Nest tree characteristics in California have been defined by Lehman (1980) as being 41 to 46 inches in diameter at breast height and in excess of 100 feet tall.

In California, 73 percent of the nest sites were within 0.5 mile of a body of water, and 89 percent within 1 mile. No nests were known to be over 2 miles from water. Of 21 nests in Oregon, Anthony and Isaacs (1989) found 85% were within one mile of water. Bald eagles often construct several nests within a territory and alternate between them from year to year. Up to five alternative nests may be constructed within a single territory (U. S. Fish and Wildlife Service 1986).

Snags, trees with exposed lateral limbs, or trees with dead tops are often present in nesting territories and are used for perching or as points of access to and from the nest. Such trees also provide vantage points from which territories can be guarded and defended. Andrew and Mosher (1982) found that successful nests were in denser forest stands farther from human disturbance than were unsuccessful ones. They identify the most important characteristics of bald eagle nesting habitat in the Chesapeake Bay as being close to water and having open mature vegetation structure that allows for easy flight.

Breeding is initiated as early as January 1 via courtship, pair bonding, and territory establishment, and normally ends approximately August 31, as the fledglings are no longer attached to the immediate nest site. This time frame may vary with local conditions and knowledge. Incubation may begin in late February to mid-March, with the nestling period extending to as late as the end of June. From June through August, the fledglings remain restricted to the nest until they are able to move around within their environment.

Anthony and Isaacs (1985) found negative relationships between eagle productivity and human activities, particularly logging activities. Effective breeding area management should avoid a flight response that is typically induced by disturbance at 200 to 300 m (Grubb et al. 1992). In their study of breeding bald eagle responses to human activities, Grubb et al. (1992) recommend a no activity primary zone of 500 to 600 m (1640 to 1968 feet) from nest sites, followed by a secondary zone of 1000 to 1200 m (3280 to 3936 feet).

Wintering habitat is associated with open bodies of water, primarily in the Klamath Basin (Dietrich 1981, 1982). Smaller concentrations of wintering birds are found at most of the larger lakes, at man-made reservoirs in the mountainous interior of the north half of the state, and at scattered reservoirs in central and southwestern California. Some of the state's breeding eagles winter near their nesting territories.

In southwestern National Forests, Grubb and Kennedy (1982) found that although live ponderosa pine trees were the most prevalent perch trees available to eagles, they preferred to use snags instead of living trees. Use of a perch tree relates to the habitat that surrounds it. Perches were oriented to provide all of the following, but not necessarily all at the same time: (1) a good view of the adjacent water and surrounding area; (2) maximum exposure to the sun, especially during morning hours on cold days; (3) maximum benefit of topography and diurnal wind currents for flight. They found eagles selecting for perches that provide good visibility, and this is influenced by three interrelated characteristics: openness, height of the substrate, and the height of the surrounding vegetation. As foliar density of the surrounding vegetation increased, or the height of the vegetation or hill increased, so did the need for higher perches. Usually eagles chose the largest trees with suitable branches.

Habitat requirements for communal night roosting are different from those for diurnal perching. Communal roosts are invariably near a rich food resource. In forest stands that are uneven-aged, communal roosts have at least a remnant of old-growth forest components (Anthony et al. 1982). Most communal winter roosts used by bald eagles throughout the Pacific recovery areas offer considerably more protection from the weather than diurnal habitat. Of three night roosts studied in southwestern National Forests, all were in ponderosa pine stands several hundred yards to several miles from the daytime water resource (Grubb and Kennedy 1982). Most roost trees were living and well foliated, but with large "windows" in the canopy. In five communal roosts in the Klamath Basin, Keister and Anthony (1983) found that bald eagles used old-growth forest stands as far as 9.6 miles from the food source. Defoliated trees such as snags, spike-topped conifers, and large deciduous trees were especially preferred.

The most common food sources for bald eagle in the Pacific region are fish, waterfowl, jackrabbits, and various types of carrion (USFWS 1986). In the winter, major prey may include:

waterfowl, ungulate carrion, and small mammalian prey (Grubb and Kennedy 1982, Grubb 1995b). The kinds of prey selected changes depending on its availability.

Many studies show that eagles avoid or are adversely affected by human disturbance (Stalmaster and Newman 1978, Andrew and Mosher 1982, Fraser 1985, Fraser et al. 1985, Knight and Skagen 1987, Buehler et al. 1991, Grubb and King 1991, Grubb et al. 1992, Chandler et al. 1995, Grubb et al. 1995, Mathisen et al. 1997). Disturbance is most critical during: nest building, courtship, egg laying and incubation (Dietrich 1990). Grubb et al. (1992) found that eagles are disturbed by most activities that occur within 1500 feet; and they take flight when activities occur within 600 feet. Mathisen et al. (1997) recommend that managers avoid any activities within 500 to 600 meters (1640 to 1968 feet) from a nest. They also recommend that any activities occurring within a secondary zone of 1000 to 1200 meters (3280 to 3936 feet) minimize the duration of the disturbance and avoid causing a flight response.

Eagles are disturbed differently depending on the kind of disturbance, the noise that it creates, the length of time that it lasts, and its location. Eagles are more disturbed as noise levels increase, the source of the disturbance gets closer, and by unusual disturbances not normally occurring in a particular area. Grubb and King (1991) and Grubb et al. (1992) found that pedestrian activities were the most disturbing group of human activities, followed by boats and vehicles. Among aircraft, helicopters elicited the highest disturbance response from eagles, frequently causing them to fly. They recommend permitting only short duration flights within 1100 m (3600 ft) of a nest (Grubb and King 1991), and they found that a greater frequency of disturbances appeared to have a greater effect on breeding eagles (Grubb et al. 1992). Position is also important, with activities located above an eagle being more disturbing than below.

Within the Tahoe National Forest, twelve breeding territories have been identified within the forest boundary. Seven nest territories are on National Forest System land (2 at Stampede Reservoir, 1 at Boca Reservoir, Independence Lake, Prosser Reservoir and Deer Creek, and 4 at Bullards Bar Reservoir). Four nesting territories on private land occur within the forest boundary; one each at Fordyce Reservoir, Webber Lake, Spaulding Reservoir, and south of Milton Reservoir, and there is one nesting territory on State land at Donner Lake. Meadow Lake had fledglings in 2002 but no nest was located.

The Tahoe National Forest lies within Zone 28 (Sierra-Nevada Mountains) of the Pacific Bald Eagle Recovery Area (USFWS 1986, p.138). Recovery goals identify a target of six territories on the forest, three territories at Bullards Bar Reservoir, and one territory each for Stampede, Boca, and Jackson Meadows. Considering the previously mentioned twelve territories within the Tahoe National Forest (assuming the Milton Reservoir territory substitutes for Jackson because of its close proximity), recovery goals for the numbers of territories have been met.

Potential risk factors to the bald eagle from resource management activities include modification or loss of habitat or habitat components (primarily large trees) and behavioral disturbance to nesting eagles from vegetation treatment, facilities maintenance, recreation, or other associated activities within occupied habitat, which could prevent or inhibit nesting or lead to nest failure (USDA Forest Service 2001).

The main stems of Lavezzola Creek and the Downie River provide suitable foraging habitat for bald eagles. Sightings are known along the Downie River, approximately 3 miles above Downieville. Most bald eagle nests in California are associated with Reservoirs, and all nests in the Tahoe National Forest are associated with Reservoirs. The District has conducted mid-winter surveys for bald eagles along the North Yuba River for 30 years, and no nesting territories have been found. The closest bald eagle nest is located 15 miles west, where the North Yuba flows into Bullards Reservoir. Trail proposals were buffered by 0.25 miles, which would include potential noise disturbances to bald eagles from trail construction or people using the trails. The proposed actions do not occur within or in proximity to suitable bald eagle nesting or foraging habitat.

B. Bald Eagle: Effects of the Proposed Action and Alternatives including Project Design Standards

Due to the lack of suitable habitat, this project will not affect this species or its habitat, and no further analysis is needed.

C. Bald Eagle: Conclusion and Determination

It is my determination that implementation of this project will not affect the bald eagle.

CALIFORNIA SPOTTED OWL

Status: USFS R5 Sensitive

A. California Spotted Owl: Existing Environment

The California spotted owl is on the USFS R5 Sensitive Species List for the Tahoe National Forest and is a Management Indicator Species on all National Forests in the Sierra Nevada Bioregion. There are three subspecies of spotted owls: the California spotted owl, the northern spotted owl, and the Mexican spotted owl. Both the northern and Mexican subspecies are listed as Threatened by the USFWS. The three subspecies occupy fairly geographically distinct areas, with the California spotted owl in the southern Cascades south throughout the Sierra Nevada mountains, the mountainous regions of southern California, and the central coast ranges at least as far north as Monterey County (Gutiérrez and Barrowclough 2005). The elevation of known nest sites range from about 1,000 feet to 7,700 feet, with about 86 percent occurring between 3,000 and 7,000 feet. The California spotted owl was petitioned for listing as Threatened or Endangered, but upon status review the USFWS found it did not warranted listing on May 24, 2006 (USFWS 2003; 68 FR 7580, USFWS 2006; 71 FR 29886). The USFWS was again petitioned to list the California spotted owl in 2015 and issued a 90-day finding on September 17, 2015 that found the petition contained information to warrant a more in-depth review of the species' conservation status (USFWS 2015; 80 FR 56423).

The Forest Plan, as amended in 2004, includes conservation strategies to maintain environmental conditions and habitat for old forest associated species in the Sierra Nevada, particularly the

California spotted owl. The strategy seeks to maintain canopy cover, large trees, and other components such as logs and snags that are known to be important to California spotted owls, while addressing the need to reduce the threat of stand-replacing wildfires to owl habitat (USDA Forest Service 2004). Although spotted owls are known to use burned forest after fires, burned forest habitat is short-lived, and is typically replaced by unsuitable shrubs or small trees within a decade, and can take over a century to return to mature forest conditions.

In February 2003, the Tahoe National Forest refined existing PACs (Protected Activity Centers) and HRCAs (Home Range Core Areas) and delineated new ones according to direction in the SNFPA (USDA Forest Service 2001). This work is updated at least once a year to add new, or revise boundaries of existing PACs and HRCAs. There are approximately 204,756 acres included within approximately 199 PAC/HRCAs in the Tahoe National Forest. Surveys for the California spotted owl have been conducted in the Tahoe National Forest since the late 1970s; some territories are monitored yearly as part of a regional demography study. Surveys conducted in the Tahoe National Forest follow the Pacific Southwest Region Protocol for Surveying for Spotted Owls in Proposed Management Activity Areas and Habitat Conservation Areas (USDA Forest Service, March 12, 1991, revised February 1993).

The Tahoe National Forest includes one of the nine geographic areas of concern identified in the CASPO report (Beck and Gould 1992). This area of concern approximately incorporates the middle third of the forest, and was identified because the checkerboard pattern of public and private lands increases the uncertainty that owl habitat would be maintained across ownerships (Beck and Gould 1992). When combined with the natural habitat fragmentation of the higher elevation territories by rock outcrops and the resulting relatively low spotted owl density, landscape-scale habitat fragmentation could occur from east to west. This increases the risk to owl populations if the owl's status in the Sierra Nevada deteriorates (Beck and Gould 1992).

California spotted owls utilize Sierra mixed conifer, ponderosa pine, red fir and montane hardwood forest types with high structural diversity, and dominated by medium (12-24") and large (>24") trees and with moderate to high levels of canopy cover (generally >40%) (Bias and Gutiérrez 1992, Call et al. 1992, Gutiérrez et al.1992, Verner et al. 1992b, Zabel et al. 1992, Moen and Gutiérrez 1997, Blakesley 2003, Blakesley et al. 2005, Chatfield 2005, Lee and Irwin 2005, Seamans 2005). California spotted owl habitat preference has long been tied to high canopy cover particularly in nest stands. Tempel et al. (2014) found a strong correlation of site occupancy with mid and late seral mixed-conifer forests with very high canopy cover (>70%). In a recent analysis (North et al. 2017) using newly acquired, highly accurate vegetation data (LiDAR), tall trees (>48m) were found to have the highest correlation with site selection while cover in the lower canopy (2-16m) was avoided. Although high canopy cover mostly occurs where tall trees are dominant, locations with high canopy cover that does not contain large trees is not tied to nest sites or PACs (North et al. 2017).

California spotted owl habitat can be assessed at multiple scales. On the Eldorado study area, Chatfield (2005) modeled habitat with circular plots centered on owl nest and/or roost locations of approximately 100, 300, and 1,170 acres, representing the nest stand, PAC, and territory scales, respectively. Seamans (2005) analyzed owls on the Eldorado study area and defined a territory as a circle with radius half the mean nearest neighbor distance of occupied territories, resulting in a circle encompassing 988 acres. Seamans (2005) found that this territory size (988 acres) encompassed >90% of all known roosts. Berigan et al (2012) found that the 300-acre PAC of the best, most mature habitat around activity centers appeared to account for much of the areas used by owls during the nesting season.

Spotted owl home range sizes are extremely variable across their range, and are suspected to be linked to availability of prey (Verner et al. 1992b, Zabel et al. 1992, Zabel et al. 1995, Bingham and Noon 1997). Bingham and Noon (1997) found that home range sizes of California spotted owls on Lassen National Forest (n = 4) averaged 6-8 times larger than estimates for northern spotted owls (n = 20) and noted that this is believed to reflect differences in habitat composition and prey availability rather than subspecific differences. California spotted owl home range is smallest in habitats at relatively low elevations that are dominated by hardwoods, intermediate in size in mixed-conifer forests, and largest in true fir forests (Zabel et al. 1992). At the time of the CASPO report in 1992, in the Sierra conifer forests a rough estimate of mean home range for California spotted owl pairs based on available information was 4,200 acres (Zabel et al. 1992).

Nesting habitat is primarily dominated by medium (12-24” dbh) to large (>24”) trees and multi-storied stands with dense canopy closure (generally >70%) (Verner et al. 1992b, Moen and Gutiérrez 1997, North et al. 2000, Blakesley 2003, Blakesley et al. 2005). Nests can be found in side cavities of live and dead firs and pines, in the top of broken-topped trees and snags, in platform nests which naturally exist in branching structures or which were built by another species, or in mistletoe brooms (Gutiérrez et al. 1992, Blakesley et al. 2005). Blakesley et al. (2005) found the mean diameter of nest trees on the Lassen study area was 46” dbh, with over 90% of nests in >30” dbh trees. Large remnant trees (>30”), even if they occur at low density (<0.5/acre), appear important to serve as nest trees (Blakesley 2003, Blakesley et al. 2005). Large trees typically provide tall, dense, canopies with open understories, suitable nesting cavities, and structural complexity, which benefits prey species for foraging and nesting. Bond et al. (2004) found the number of large trees (>30”) and higher canopy cover to be the most important habitat variables in defining nesting habitat. North et al. 2017 found that gaps (112-1000m$^{2)}$ were rare in nest stands, and infrequent (.17-1.21%) in PACs or territories.

Foraging habitat includes mid- to late-seral forest with at least 40-50% canopy closure (Verner et al. 1992b). Irwin et al. (2007) found optimal foraging habitat consisted of moderately-dense forest with basal area from 152 to 240 ft^2/acre in Douglas-fir, white fir, and red fir, and greater basal area of large (>8” dbh) hardwoods. Similarly Irwin et al. (2015) found areas that were harvested and maintained 202 to 282 ft^2/acre of midstory basal area had the highest likelihood of being used. Daytime roosts are typically in denser forests with greater basal area and overstory canopy cover than for nocturnal roosts (Irwin et al. 2007).

Spotted owl populations exhibit high adult survival (>80%) but highly variable reproduction and recruitment (Blakesley et al. 2001, Seamans 2005, Blakesley et al. 2006a). The spotted owl population growth rate appears to be most dependent on adult survival (Lande 1988, Noon and Biles 1990, Blakesley et al. 2001, Seamans 2005, Blakesley et al. 2006a). While adult survival is the most important variable to the population growth rate, annual variability in the population growth rate is influenced by reproductive output and juvenile survival (Seamans 2005). There is a high level of annual variation in the proportion of pairs that nest, and a high level of annual variation in nesting success (Noon and Biles 1990, Verner et al. 1992, North et al. 2000, Blakesley et al. 2001, USDA Forest Service 2009). Blakesley et al. (2005) found that nest success was higher when large remnant trees (>30") were present in the nest stand, and higher in nest stands dominated by medium sized trees (12-24") than in stands dominated by large trees (>24"). Temple et al. (2016) found that edge habitat within the territory is negatively associated with occupancy.

Nesting attempts and nesting success appear to be connected to abiotic environmental factors, especially the weather (North et al. 2000, Lee and Irwin 2005, Seamans 2005, USDA Forest Service 2009). In the southern Sierra Nevada, North et al. (2000) noted that within any given year in the southern Sierra Nevada reproductive success was largely synchronous among all owl pairs, negatively correlated with nesting period precipitation in oak woodlands and conifer forest, and positively correlated with April's minimum temperature in conifer forests. Reproductive output was also correlated with high levels of foliage volumes over the nest, suggesting this provides protection from precipitation (North et al. 2000). Seamans (2005) modeled demographic parameters of 15 years of data on the Eldorado study area, and found that the top model suggested that reproductive output was negatively correlated with cold and wet conditions during incubation. LaHaye et al. (2004) found that fecundity was lower during wet spring seasons, and increased with increasing precipitation during the previous year. Prey availability, also subject to the effects of weather, can have major effects on general owl biology such as reproductive rates, timing and location of nesting, whether nesting occurs, density of nesting pairs, and dispersal or major movements of whole populations (Verner et al. 1992b).

California spotted owls have strong site fidelity and establish a strong pair bond (Blakesley et al. 2006b). They do not stay together, however, during the non-nesting season (Verner et al. 1992b). They exhibit individual variation in migratory behavior; in the non-nesting season any particular owl may migrate to lower elevations, stay in the same general area used during the nesting season, or move back and forth between areas (Verner et al. 1992b). Spotted owls use similar habitats in the nesting and non-nesting seasons (Irwin et al. 2007). Approximately 7% of adult California spotted owls change territories during the non-breeding season; these movement generally consists of younger owls, single owls, paired owls that lost their mates, owls at lower quality sites, and owls that failed to reproduce in the prior year (Blakesley et al. 2006b).

The northern flying squirrel (*Glaucomys sabrinus*) and dusky-footed woodrat (*Neotoma fuscipes*) comprise the two primary prey species of the California spotted owl, with the flying squirrel the predominate prey in the higher elevation conifer forest and the woodrat the predominate prey in the lower elevation forests and woodlands (Williams et al. 1992, Munton et al. 2002, USDA Forest Service 2009). Pocket gophers (*Thomomys* spp.) were the second largest component (in biomass) of owl diets on Sierra National Forest in both the higher conifer-dominated elevations and the lower woodland elevations (Munton et al. 2002). Other prey items are other small mammals (especially *Peromyscus* spp.), birds, lizards, and insects (Munton et al. 2002, USDA Forest Service 2009).

Risk factors for the California spotted owl include loss of habitat, habitat fragmentation, reduction in habitat quality, climate change, the effects of wildfire, disturbance at breeding sites, the invasive barred owl, disease, and blood parasites (USDA Forest Service 2001a, Vol. 3, pp. 69-112, Ishak et al. 2008, USDA Forest Service 2009).

The invasive and larger barred owl poses a threat to the California spotted owl due to competition for food and nesting resources, possible displacement, hybridization with the spotted owl, and potentially increased spread of disease and blood parasites (Ishak et al. 2008, USDA Forest Service 2009). Beginning around the late 1800s, the barred owl expanded its range from the forests east of the Great Plains to forests in the western United States, arriving in northern California around the late 1970s from Oregon, and in the Sierra Nevada in the 1980s where they have continued to increase in abundance though at a slower rate than their expansion in Washington and Oregon (Livezey 2008b, USDA Forest Service 2009). The barred owl is more of a habitat generalist than the spotted owl, occupying a greater variety of habitats and having a wider range of prey than the spotted owl (Livezey 2007, Livezey et al. 2008a). There is one known barred owl territory on the Yuba River Ranger District on the Tahoe National Forest which has existed since the early 1980s. Surveys in 2016 detected a barred owl on the American River District of the Tahoe National Forest.

Disturbance to California spotted owls is another potential risk factor. Wasser et al. (1997) measured significantly higher levels of stress hormones in male northern spotted owls whose home range centers were within 0.41 km (0.25 mi.) of major logging roads or recent (10 years to present) timber activity. Hayward et al. (2011) found that northern spotted owls that nested near noisy roads fledged fewer young then those exposed to less noise, indicating that noise exposure may effect reproductive success. Forest Service recommendations for reducing direct effects to spotted owls have generally included minimizing disturbances within 0.25 miles of known roosts or nests during the breeding season (March 1 through August 15).

Lee and Irwin (2005) examined the potential long-term effects to California spotted owl occupancy and reproduction by landscape-level reductions in canopy cover through various combinations of mechanical forest thinning and wildland fire through six decades. They modeled various long-term scenarios of no treatment, light thinning with prescribed fire, heavy mechanical thinning, mixed-lethal fire (6 foot flame lengths), and lethal fire within spotted owl territories. The light and heavy thinning prescriptions were modeled to leave the larger trees regardless of species. Three categories of spotted owl territories were examined over a projected six decade time period, representing territories with a higher proportion of sparse canopy cover (non-reproductive territories), territories with an intermediate mix of canopy cover classes, and territories with a larger proportion of dense canopy cover. Lee and Irwin (2005) state:

> "The general trend for all scenarios except immediate lethal fire was towards higher proportions of intermediate canopy cover (40-69%) and lower proportions of sparse canopy cover (0-39%). The mechanical thinning and mechanical thinning plus DFPZ construction scenarios resulted in less of the dense canopy class (70-100%), but equal or more amounts of intermediate canopy levels than the let-grow scenario through time. Mixed-lethal fire produced a pronounced effect in the decade that the simulated fire occurred (the second decade), which was still discernible 4 decades later. None of the simulated trajectories moved beyond the range of observed variation in the original data, suggesting that expected effects on owl reproduction would be essentially immeasurable.

> Our simulation results lend credence to the hypothesis that modest fuels treatments are compatible with territory-level canopy cover needs for spotted owl reproduction in the Sierra Nevada."

Lee and Irwin (2005) note that their analysis of fire effects was simplified when compared to the complex fire behavior characteristics of most landscapes, and that all potentially complex habitat elements important to spotted owls were not analyzed. They specify that the entire complex of factors affecting owls should be considered when designing and implementing thinning projects in order to minimize risk to spotted owls.

A recent analysis from the nearby Eldorado study area population concluded that California spotted owl occupancy declined 31% and the population declined 50% between 1990 and 2012 (Tempel et al. 2014). An analysis of the effects of forest management on California spotted owls using data from the same study concluded that reductions in canopy cover in dense stands (>70% canopy cover) from either logging or wildfire may be contributing to the study population decline (Tempel et al. 2014). That study noted that the actual effect of medium-intensity timber harvests on reproduction was only weakly supported (ibid.). Tempel et al. (2016) concluded "Our results suggest that some fuels treatments intended to reduce fire risk and improve forest resilience could be located within Spotted Owl territories without adversely impacting territory occupancy if such treatments do not consistently reduce canopy cover below 50%". Irwin et al. (2015) found that spotted owls use foraging habitat that has been harvested, equally or more often then prior to harvest. Irwin et al. (2015) noted that foraging post harvest was most likely to occur in stand that maintained 202–282ft^2/acre of mid story basal area. Jones et al. 2017 found that spotted old declines seemed to be linked to an extinction debt from historic logging of large trees. An "extinction debt" is caused by initial survival after habitat modification but leads to continued declines until a new equilibrium is reached (Jones et al. 2017).

Bond et al. (2009) found that a mosaic of burn severities in a wildfire, like that which occurred in the McNally Fire on Sequoia National Forest in 2002, through previously established California spotted owl territories has been found to maintain spotted owls in the area for at least four years post-fire. The McNally Fire of 2002 burned with variable severity creating a mosaic across the landscape; of the conifer forest within the fire perimeter, 31% remained unburned, 29% burned at low severity, 27% burned at moderate severity, and 13% burned at high severity (Bond et al. 2009). In their study, low severity burned areas were preferentially selected for roost sites, and high severity burned areas were preferentially selected for foraging over unburned sites (Bond et al. 2009). Preferential use of the burned areas for foraging in this mosaic may be due to the increased shrub and forb understory, and accompanying increased prey availability as a result (Lee and Tietje 2005, Innes et al. 2007, Bond et al. 2009). In a recent study of the 2014 King Fire, Jones et al. (2016a) found that nine spotted owls fitted with radio transmitters diffinitively avoided areas that had burned at high severity. Jones et al. (2016a) found that areas that burned at low severity and unburned forest was used proportionally to their availability on the landscape. A recent analysis (Stephens et al. 2016) concluded that if wildfire trends continued, the majority of California spotted owl nesting habitat may be considerably altered by fire within the next 75 years.

Spotted owl surveys were conducted for all trail proposals that occur within 0.25 miles of spotted owl habitat in 2017, and a second year of surveys conducted in 2018. This distance was selected to address potential effects of directly removing spotted owl nesting or roosting habitat, or effects from noise disturbances that could occur to owls from trail construction, relocation, decommissioning, or in the future from use of the trails over the long term. Surveys were focused on determining whether historic PACs were occupied, and to cover moderate to high quality habitat that might be affected by project proposals. Portions of the Pauley trail proposed for decommissioning and re-route lies within a spotted owl Protected Activity Center. The location of the newly re-routed trail was planned to move the trail further away from an owl activity center, and to avoid the highest quality owl habitat, with portions occurring outside of suitable habitat altogether.

To date, no spotted owl nests have been found to be located within 0.25 miles of any trail proposals. One owl responded once at night in the vicinity of the Lavezzola Trail re-route, but it was not located during the follow-up visit.

B. California Spotted Owl: Effects of the Proposed Action and Alternatives including Project Design Standards

The following Management Requirements are included in the proposed action, would reduce potential impacts to spotted owls:

To avoid disturbances to spotted owls during the breeding season, Limit the Operating Period from March 1 through August 15 for trail construction and decommissioning along the Lavezzola Trail re-route, unless two-year surveys are completed to protocol, and it is determined that this is no longer needed.

Implement project-specific Best Management Practices include measures to minimize sedimentation into creeks and maintain water quality.

Direct and Indirect Effects

Owls could be disturbed and displaced from roosts or nests, if they are located in proximity to locations where trails would be decommissioned or re-located. Surveys were conducted in 2017, and no spotted owl nests or roosts have been located that occur within 0.25 miles of project-related activities. However, second-year surveys would need to be conducted in 2018, before it could be determined that owls are not likely to be nesting within suitable habitat. The project Management Requirements limit the operating season for trail construction and decommissioning along the Lavezzola Trail where it occurs within suitable habitat, to insure that nesting owls would not be disturbed during the breeding season.

Table 2 identifies potentially affected spotted owl territories where either trail re-routes or obliteration is proposed within spotted owl PACs or HRCAs, showing whether it is a re-routed trail, or obliteratation. New trail would remove less than 1 acre of understory habitat, and trail obliteration would restore less than 1 acre. Noise disturbance from use of the re-routed trails would extend out approximately 0.25 miles. No spotted owls have been found to be nesting or roosting within this distance. The trails would be used during the day, while spotted owls are active at nights. Therefore, longer term noise disturbances to spotted owls are not expected to occur from use of the new trails, because spotted owls would not be disturbed during the nesting season, or while they are out foraging at night.

Table 2. Spotted owl territories, where trail proposals overlap portions of their Protected Activity Centers or Home Range Core Areas, showing the miles and acres of trail that represents new re-routes compared with obliteration.

	Trail re-route		**Total Re-route**	**Trail Obliteration**		**Total Obliterated**
	PAC	**HRCA**	**PAC+HRCA**	**PAC**	**HRCA**	
SIE118 Big Boulder	0.23 mi.	0.57 mi.	0.8	0.64	0.16	0.8
SIE 049 Big Grizzly	0	0.24	0.24	0	0.17	0.17
SIE 050 Rattlesnake	0	0.39	0.39	0	0.04	0.04
SIE 052 Lavezzola	0.18	3.55	3.73	0	0.91	0.91
Total mi.	**0.41**	**4.75**	**5.16**	**0.64**	**1.43**	**1.92**
Total Ac.	**0.1 ac**	**0.5 ac.**	**0.6 ac.**	**0.1 ac**	**0.2**	**0.3**

Most trail construction is designed to avoid the need to remove large trees, snags, and large downed wood. However, to maintain a desirable trail grade that minimizes soil erosion, some trees and hazardous snags may need to be removed. Any trees felled would be retained and recruited as downed logs, with only a four foot wide trail prism constructed. This project would result in a net decrease of trails within special management areas for the California spotted owl than currently exists, while it addresses excessive soil erosion that is occurring from overly steep trails in the project area.

Cumulative

The cumulative effects area includes all of the trail proposals, buffered by 0.25 miles. This area would account for potential effects of directly removing spotted owl nesting or roosting habitat, or effects from noise disturbances that could occur to owls from trail construction, relocation, decommissioning, or in the future from use of the trails over the long term. There is no private land that overlaps with this project area. Cumulative effects to owls within the project area occur primarily noise disturbances from recreational use along of the trails, dispersed recreational use off of roads and trails, and road and trail maintenance to manage encroaching vegetation and fell hazard trees. Because the majority of the project lies within the East-West Yuba RARE II area, there is little additional vegetation management that is conducted. There are no additional projects that are planned within the project area. This project would result in a net decrease of trails that would occur within spotted owl management areas, which would improve habitat quality for owls over time by decrease the degree of human disturbances to owls in the project area.

C. California Spotted Owl: Conclusion and Determination

It is my determination that implementation of Alternative __ may affect individuals, but is not likely to result in a trend toward Federal listing or loss of viability for the California spotted owl within the planning area of the Tahoe National Forest. In the absence of a range wide viability assessment, this viability determination is based on local knowledge of the California spotted owl as discussed previously in this evaluation, and professional judgment.

GREAT GRAY OWL

Status: USFS R5 Sensitive

A. Great Gray Owl: Existing Environment

The distribution of the great gray owl is circumboreal, with the Sierra Nevada encompassing the most southern extent of the species (Beck and Winter 2000). The core range of the great gray owl in California is centered on the greater Yosemite National Park area with an estimated population in California totaling fewer then 300 individuals (Winter 1986, Greene 1995, Beck and Winter 2000, Sears 2006, Wu et al. 2015). The Yosemite population has been identified as a genetically distict population and taxon, *Strix nebulosa yosemitensis* (Hull et al. 2015). There are records of great gray owls as far south as Tulare County, and to the north from the Modoc, Lassen, Plumas, Tahoe, and Eldorado National Forests, and from Del Norte, Humboldt, Shasta, and Siskiyou Counties (Beck and Winter 2000).

Until recently great gray owls were thought to require two particular habitat components; a meadow system with a sufficient prey base, and adjoining forest with adequate cover and nesting structures (Winter 1980, Winter 1986, Greene 1995, van Riper and van Wagtendonk 2006). Meadows appear to be important foraging habitat for great gray owls, where approximately 93% of their prey is taken (Winter 1981). Data from the greater Yosemite area suggests that to support persistent occupancy and reproduction, meadows need to be at least 25 acres, but meadows as small as 10 acres may support infrequent breeding (Winter 1986, Beck and Winter 2000). Using radio telemetry, van Riper and van Wagtendonk (2006) found that over 60% of all great gray owl locations were within 100 meters (328 feet) of a meadow; 80% of locations were within 200 meters (656 feet) of a meadow. Recently nesting great gray owls in the western Sierra Nevada have been found to be associated with openings other than meadows in lower elevations where oak woodlands transition to coniferous forests (Polasik et al. 2016). Polasik et al. (2016) documented owls nesting in large diameter trees with the landscape dominated by smaller trees, interspersed with annual grasslands. Recent research has also found that great gray owls have nested at lower elevations, and farther distances from meadows and opening then previously considered suitable (Wu et al. 2015). Wu et al. (2015) found that 21% of the nest they monitored were below 1,000m and 31% of the nests were >750m from a meadow. Great gray owls have also been documented in northeastern Oregon foraging in open forest, clear-cuts, and burned areas where these areas support a high cover (eg. mean 88% in forest with canopy cover 11-59%) grass-forb habitat (Bull and Henjum 1990). In comparing number of large snags (>24" dbh), smaller snags (<24" dbh), and canopy cover, Greene (1995) found that high canopy cover was the only variable significantly higher in occupied habitat, possibly due to its effect on microclimate.

In the Sierra Nevada, great gray owl breeding activity is generally found in mixed coniferous forest from 2,500 to 8,000 feet elevation where such forests occur in combination with meadows or other vegetated openings (Greene 1995, Beck and Winter 2000), but have recently nested as low as 2,300 feet (Hull et al. 2014). In their study in Yosemite National Park, van Riper and van Wagtendonk (2006) found that home ranges were located adjacent to meadows in red fir and Sierra mixed conifer most frequently, and home range boundaries followed meadow and drainage topography. They found that most females nested where red fir was the most common habitat type, and some nested in habitat dominated by lodgepole pine (van Riper and van Wagtendonk 2006). Habitat types used by breeding females included Sierra mixed conifer, montane riparian, and montane chaparral types (van Riper and van Wagtendonk 2006). Nesting usually occurs within 840 feet (averaging 500 feet) of the forest edge and adjacent open foraging habitat (Beck and Winter 2000). Greene (1995) found that nest sites had greater canopy closure (mean 84%) and were more likely located on northern aspects than expected by chance.

As with most owls, great gray owls do not build their nests (Bull and Henjum 1990, Greene 1995). In contrast to northeastern Oregon and elsewhere where platforms such as old hawk nests and mistletoe infected limbs are the predominant nest substrate (Bull and Henjum 1990), most nests in the Sierra Nevada have generally been found at the top of large broken top fir snags; fir snags tend to break at right angles and form a suitable nest substrate (Winter 1986, Greene 1995). Greene (1995) found that the next most preferred species for nesting was black oak, found at the lower elevations. Greene (1995) found that nest trees in Stanislaus National Forest averaged 32 inches dbh and 32 feet tall, while those in Yosemite National Park averaged 44 inches dbh and 45 feet tall. In northeastern Oregon, Bull and Henjum (1990) found that great gray owls readily used artificial open platforms, preferentially 49 feet high but also 30 feet high if none higher were available, and preferentially in closed forested stands versus those adjacent to clear-cuts.

In the Yosemite area, males begin establishing nesting territories in March to early April (Beck 1985). After 30 to 36 days of incubation, eggs hatch from mid-May to mid-June. Young begin to fledge in early June to early July, but will remain around the nest through August. However, great gray owls will breed earlier at higher elevations (approximately 2 weeks earlier for every 1,000 foot increase in elevation).

In Yosemite National Park, van Riper and van Wagtendonk (2006) found breeding season home ranges (95% adaptive kernel) averaged 152 acres for females and 49 acres for males. Breeding adults were found to utilize specific activity centers within the home range, with radio telemetry locations densely packed in localized areas; the activity centers averaged 43 acres (based on the 75% adaptive kernel home range). Activity centers were based around nests or roost sites but also included nearby foraging areas, and were frequently associated with outer meadow boundaries (van Riper and van Wagtendonk 2006). While much larger than breeding season home ranges, non-breeding season home ranges were also centered on wet meadows (van Riper and van Wagtendonk 2006). During the non-breeding season, home ranges averaged 6,072 acres for females and 5,221 acres for males (van Riper and van Wagtendonk 2006).

Great gray owls hunt by perching at the edges of meadows or grasslands and listening for prey in grass runways or underground burrows, then flying low over the ground and dropping on the prey (Brunton 1971, Nero 1969, Winter 1981). During the majority of the breeding season, males do a majority of the hunting, often by day, and provide food to the nest (Greene 1995). Larger trees possibly have more open limb development, allowing stooping and less view obstruction. Winter (1982) also observed owls using fence posts as hunting perches. Greene (1995) found that hunting perches were generally located in drier microsites. The lack of perches at the edges and/or within meadows may render a meadow unsuitable for great gray owls.

Prey of great gray owls is primarily pocket gophers and voles (Winter 1986, Reid 1989, Bull et al. 1989). If prey numbers are low in any given year, great gray owls may occupy a site but may not nest, possibly due to the lack of ability to feed young (Bull and Henjum 1990). Greene (1995) found that occupied and reproductive great gray owl habitat corresponded more closely to greater vole than greater gopher abundance; his results suggested that gophers are generally more abundant than voles, but they are probably less available to great gray owls due to the fact they are typically underground. Pocket gophers tend to inhabit areas of deep unsaturated soils allowing for easier burrowing and tunneling (Jones and Baxter 2004), while voles tend to inhabit wet meadows with thick grass, forbs, and sedge cover (Sera and Early 2003). As increased soil moisture improves habitat for voles, it becomes less suitable as gopher habitat; in combination with typically drier conditions observed at hunting perch sites, Greene (1995) suggested that variability in soil moisture and related vegetation conditions, to support the two primary prey taxa, may provide optimal great gray owl foraging habitat in the Sierra Nevada.

In the Tahoe National Forest, there have been few recorded great gray owl sightings, and nesting has only recently been confirmed in one location on or near private land. Possible sighting and/or detection locations include Kyburz Flat (2009 single individual, nesting not confirmed), near Independence Lake (2012 a pair was detected but nesting not confirmed) Perazzo Meadows (May 2004 nesting not confirmed), along Pliocene Ridge Road (occasional sightings since 2003 with confirmed nesting in the area in 2009), three miles north of Nevada City (an adult located in January 1996 and January 1997), Donner Ranch Ski Area (pair observed in November 1994), near Spencer Lakes at the northern border of the Tahoe National Forest (detection in July 1990), south of Lincoln Creek Campground (an individual in July 1987), and near Sattley (pair in January 1985).

This project occurs outside of any meadow habitat or great gray owl habitat.

B. Great Gray Owl: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct, Indirect, and Cumulative

Due to the lack of suitable habitat within the project area, there are no direct, indirect, or cumulative effects that would occur from this project.

C. Great Gray Owl: Conclusion and Determination

It is my determination that implementation of this project not affect the great gray owl.

NORTHERN GOSHAWK

Status: USFS R5 Sensitive

A. Northern Goshawk: Existing Environment

The northern goshawk (*Accipiter gentilis)* is listed on the USFS R5 Sensitive Species List for the Tahoe National Forest. In 1997 the northern goshawk was petitioned for listing as threatened or endangered west of the 100th meridian, but upon status review the USFWS found it did not warrant listing (USFWS 1998; 63 FR 35183).

As prescribed by the Forest Plan, surveys are conducted in compliance with the Pacific Southwest Region's survey protocols during the planning process when vegetation treatments are likely to reduce habitat quality are proposed in suitable northern goshawk nesting habitat that is not within an existing California spotted owl or northern goshawk PAC (USDA Forest Service 2000).

Standards and guidelines for northern goshawk management are defined in the Forest Plan, as amended. Current guidelines include delineation of protected activity centers (PACs) surrounding all known and newly discovered breeding territories detected on National Forest System lands. Northern goshawk PACs are designated based upon the latest documented nest site and location(s) of alternate nests. If the actual nest site is not located, the PAC is designated based on the location of territorial adult birds or recently fledged juvenile goshawks during the fledgling dependency period (USDA Forest Service 2004).

Northern goshawk PACs are delineated to: (1) include known and suspected nest stands and (2) encompass the best available 200 acres of forested habitat in the largest contiguous patches possible, based on aerial photography. Where suitable nesting habitat occurs in small patches, PACs are defined as multiple blocks in the largest best available patches within 0.5 miles of one another. Best available forested stands for PACs have the following characteristics: (1) trees in the dominant and co-dominant crown classes average 24 inches dbh or greater; (2) in westside conifer and eastside mixed conifer forest types, stands have at least 70 percent tree canopy cover; and (3) in eastside pine forest types, stands have at least 60 percent tree canopy cover. Non-forest vegetation (such as brush and meadows) should not be counted as part of the 200 acres. As additional nest location and habitat data become available, PAC boundaries are reviewed and adjusted as necessary to better include known and suspected nest stands and to encompass the best available 200 acres of forested habitat. When activities are planned adjacent to non-national forest lands, available databases are checked for the presence of nearby northern goshawk activity centers on non-national forest lands. A 200-acre circular area, centered on the activity center, is delineated. Any part of the circular 200-acre area that lies on national forest lands is designated and managed as a northern goshawk PAC. PACs are maintained regardless of northern goshawk occupancy status. PACs may be removed from the network after a stand-replacing event if the habitat has been rendered unsuitable and there are no opportunities for re-mapping the PAC nearby.

In 1999, prior to PAC-delineation guidelines set forth in the 2001 and 2004 SNFPA, 64 Goshawk Management Areas had been identified in Tahoe National Forest based on known nest sites, territorial adults, and habitat suitability. In June 2003, the Tahoe National Forest reviewed existing Goshawk Management Area boundaries to ensure that they met the intent of the SNFPA 2001 direction for goshawk PACs. By June 2003 PAC delineation had been completed in Tahoe National Forest in accordance with SNFPA direction. At that time there were 71 northern goshawk PACs encompassing approximately 15,500 acres. Since then, additional PACs have been delineated based on new information, and as of 2014 there are 137 goshawk PACs in the Tahoe National Forest encompassing 29,427 acres.

The Northern Goshawk Scientific Committee was established in 1990 to develop a credible management strategy to conserve the goshawk in the southwestern United States. Reynolds et al. (1992) recommendations included that goshawk nesting home ranges should exist as an interspersed mosaic of various structural stages in certain proportions, and that on every acre within home ranges there should remain a few large trees in clumps that live out their lives and eventually become snags, then logs that decompose. Their recommendations focused on three components of a goshawk's nesting home range, amounting to about 6,000 acres: nest area (approximately 30 acres), post fledging-family area (PFA; approximately 420 acres), and foraging area (approximately 5,400 acres; Reynolds et al. 1992). The nest area may include more than one nest, is typically located on a northerly aspect in a drainage or canyon, and is often near a stream (Reynolds et al. 1992). Nest areas contain one or more stands of large, old trees with a dense canopy cover (Reynolds et al. 1992).

Forest types associated with goshawk nest areas vary geographically (USFWS 1998, Kennedy 2003). In the Sierra Nevada goshawks breed from the mixed conifer forests at low elevations up to and including high elevation lodge pole pine forests and eastside ponderosa pine habitats. Goshawks winter from the lodgepole pine forest down slope to blue oak savannah (Verner and Boss 1980). In the Tahoe National Forest, goshawks are year-round residents. Nests are found in all of the vegetation types listed above, as well as in aspen stands (Tahoe 1999). Andersen et al. (2005), in review of existing research on goshawks including their nesting habitat and typical high canopy closure preferences, noted that high canopy closure in relation to the range of available canopy closure may be more important for goshawk nesting than absolute canopy closure, at least above some minimum threshold.

Studies suggest that goshawks select more mature forest for nesting, with higher canopy cover and larger trees as compared to surrounding forest (e.g. Hayward and Escano 1989, Squires and Rugiero 1996, Daw and DeStefano 2001). Hypotheses for why goshawks select forest with larger trees and higher canopy cover include: 1) increased protection from predators, 2) increased food availability, 3) reduced exposure to cold temperatures and precipitation during the energetically stressful pre-egg laying period, 4) reduced exposure to high temperatures during the nestling period, 5) reduced competition with raptor species that nest in more open habitats, or 6) increased mobility because of reduced understory vegetation in mature stands (Andersen et al. 2005). Mature coniferous, mixed, and deciduous forest habitats provide large trees for nesting, a closed canopy for protection and thermal cover, and open spaces allowing maneuverability below the canopy (Fowler 1988). Analysis of vegetative data collected at 39 nest sites in the Tahoe National Forest and the Lake Tahoe Basin Management Unit indicates that goshawk nest stands in the Tahoe National Forest have a mean canopy closure of 70 percent, 32 large trees per acre (24-49 inch dbh), large amounts of dead and down logs, and slopes less than 15 percent. Research conducted on the Klamath National Forest indicated that when nest occupancy was monitored over subsequent years, re-occupancy of the nest stand was nearly 100 percent for nest clusters that were maintained at a minimum of 200 acres (Woodbridge and Dietrich 1994).

Recommendations in the Southwest Region suggest managing 5,400 acres of foraging habitat per territory (Reynolds et al. 1992). Conclusions from studies in the Sierra Nevada support similar habitat requirements (Hargis et al 1994, Keane and Morrison 1994). Important components of foraging habitat include snags (min. 3/ac. >18" dbh) and logs (min. 5/ac. >12" dbh and > 8' long) for prey base populations (USDA 1991). Management requirements for the California spotted owl are thought to provide adequate quantities of snags and down logs to support goshawk prey species in foraging habitat (Tahoe 1999). Beier and Drennan (1997) found that goshawks selected foraging sites that had higher canopy closure, greater tree density, and greater density of trees greater than 16 inches in diameter. They recommend managing stands for canopy closure values above the prescribed minimum 40%. Primary prey species include small mammals and small to medium sized birds (Verner and Boss 1980, Fowler 1988).

Goshawk nesting activities are initiated in February. Nest construction, egg laying and incubation occur through May and early June. Young birds hatch in June and begin fledging in late June and early July, and are independent by mid-September.

Potential risk factors to goshawks include effects of vegetation management and wildfire on the amount, distribution, and quality of habitat (USDA Forest Service Pacific Southwest Region 2001a). Human disturbance is also a risk factor. In the Lake Tahoe Basin Management Unit, Morrison et al. (2011) found that human activity was twice as high within infrequently as compared to frequently occupied goshawk territories, and there was a greater extent of all types of roads and trails within the infrequently occupied territories. While it is not statistically significant, Grubb et al. (1998) noted no discernible change in behavior to logging truck noise as they passed at 500 meters from two active goshawk nests. A more recent study looked at disturbance from logging trucks driving by at 78, 143, and 167 meters, which resulted in no flushing of nesting goshawks.

Moser and Garton (2009) experimentally tested the effects of clearcutting within goshawk nesting areas on reoccupancy and nesting success for two years following treatments, and found no effects on goshawk reoccupancy, nesting success, or number of fledglings between harvested and unharvested nesting areas. Their model suggested, however, that goshawk breeding area reoccupancy was a function of the amount of potential nesting habitat available in the 17-ha area surrounding the nest, with goshawks reoccupying breeding areas if they contained >39% potential nesting habitat following harvest; and that goshawks were more likely to attempt nesting after disturbance if >39% of the 170-ha (420 acres) area around their nest was left in potential nesting habitat (Moser and Garton 2009). A circular area representing 420 acres would be represented by a radius of approximately 0.4 miles.

Goshwak surveys were conducted along trail proposals and within the highest quality surrounding habitat for 0.25 miles.

B. Northern Goshawk: Effects of the Proposed Action and Alternatives including Project Design Standards

The spatial area considered for analyzing effects to the northern goshawk include all trail proposals, buffered by 0.25 miles. This area would inform potential effects from the loss of habitat or noise disturbance during trail construction, obliteration, and from use of the trail over the long term.

Project Design Standards: A management requirement is included in this project to Limit the Operating Period for trail construction and decommissioning along the Butcher Ranch Trail re-route and decommissioning, so that either activities do not occur during the goshawk breeding season (February 15 through September 15), or until surveys are completed to a two-year protocol, and it is determined that territorial goshawks are either not present, or are not nesting during the year that these activities occur.

Direct and Indirect Effects

Direct effects could occur to individuals if they are present during the time when activities occur. Surveys were conducted in 2017 to determine if any goshawk territories are located in the project area. Although no goshawk nests were found, nor was sufficient evidence obtained to identify a goshawk territory, there were positive detections of a goshawk. Consequently, this project includes a Management Requirement to limit the operating period for trail construction and decommissioning along this segment of trail. This would insure that nests are not disturbed along this segment of trail.

Cumulative Effects

The cumulative effects area includes all of the trail proposals, buffered by 0.25 miles. This area would account for potential effects of directly removing goshawk nesting habitat, or effects from noise disturbances that could occur to individuals from trail construction, relocation, decommissioning, or future use of the trails over the long term. There is no private land that overlaps with this project area. Cumulative effects to owls within the project area occur primarily noise disturbances from recreational use along of the trails, dispersed recreational use off of roads and trails, and road and trail maintenance to manage encroaching vegetation and fell hazard trees. Because the majority of the project lies within the East-West Yuba RARE II area, there is little additional vegetation management that is conducted. There are no additional projects that are planned within the project area. This project would result in a net decrease of trails that currently occur within goshawk habitat. Therefore, once this project is implemented, cumulative effects currently occurring to goshawks would decrease.

C. Northern Goshawk: Conclusion and Determination

It is my determination that implementation of this project may affect individuals, but is not likely to result in a trend toward Federal listing or loss of viability for the northern goshawk within the planning area of the Tahoe National Forest. In the absence of a range wide viability assessment, this viability determination is based on local knowledge of the northern goshawk as discussed previously in this evaluation, and professional judgment.

WILLOW FLYCATCHER

Status: USFS R5 Sensitive

A. Willow Flycatcher: Existing Environment

The willow flycatcher (*Empidonax trailii*) is listed on the USFS R5 Sensitive Species List for the Tahoe National Forest. There are three subspecies of willow flycatcher in different portions of California; they have been distinguished from each other based on distribution and color. In the Sierra Nevada, *E. t. adastus* and *E. t. brewsteri* generally occupy the eastern and western slopes, respectively; both of these subspecies likely occur in the Tahoe National Forest (Green et al. 2003). The southwestern willow flycatcher, *E. t. extimus*, occupies southern California as well as other southwestern States and was listed as endangered by the USFWS in 1995 (USFWS 1995; 60 FR 10694).

In accordance with the SNFPA Record of Decision (USDA Forest Service 2001), a Conservation Assessment of the Willow Flycatcher in the Sierra Nevada was completed (Green et al. 2003). The Conservation Assessment summarized all known and relevant information pertinent to management of the willow flycatcher in the Sierra Nevada at the time of its completion. Willow flycatcher monitoring and demographic analysis has been conducted in the central and northern Sierra Nevada since 1997 with results documenting the continued population decline (Loffland et al. 2014).

The willow flycatcher was once a common summer resident throughout California where suitable habitat existed; areas where it was most common included the Central Valley and central California in general, and the southern coastal region (Grinnell and Miller 1944, Harris et al. 1987). Observed declines in breeding populations have been a growing concern for over four decades and it is now limited to scattered meadows of the Sierra Nevada and along the Kern, Santa Margarita, and San Luis Rey Rivers (Harris et al. 1987). Loss of the willow flycatcher, especially in the Central Valley, has been largely attributed to alteration and loss of riparian habitat by various land management practices along with cowbird parasitism (Harris et al. 1987). Based on data from 14 years of the demography study, populations across the Sierra Nevada have declined 19% yearly in the Lake Tahoe and West Carson River areas to the south, and 6% yearly in the Little Truckee River area (Mathewson et al. 2012). The highest recorded number of territories on National Forest System land in the Sierra Nevada bioregion is located in the Perazzo Meadows area in the Tahoe National Forest. Systematic surveys and research on willow flycatchers have occurred throughout the Perazzo Meadow area since the early 1980s (eg. Serena 1982, Flett and Sanders 1987, Harris et al. 1987, Sanders and Flett 1989, Bombay et al. 2003, Mathewson et al. 2009).

The willow flycatcher is a small passerine Neotropical migrant that breeds during summer in riparian deciduous shrub habitat generally dominated by willows in the United States and Canada, and winters in tropical and subtropical areas from southern Mexico to northern South America (as summarized in Green et al. 2003). Willow flycatchers in the northern Sierra Nevada typically begin arriving on their breeding grounds around the 1st of June, and egg laying for first nest attempts sometimes begins as early as the second week in June, but more often in late June/early July (summarized in Green et al. 2003). Up to three nesting attempts may occur as a result of nest failure, with egg-laying through the 1st week of August, and all willow flycatchers appear to be gone from their breeding territories by mid-September (summarized in Green et al. 2003).

Breeding habitat typically includes moist meadows with perennial streams and smaller spring fed or boggy areas with willow or alders; dense thickets are generally avoided in favor of more patchy willow sites providing considerable edge (summarized in Green et al. 2003). Bombay et al. (2003) found that riparian shrub cover was the primary predictor of habitat selection at the meadow, territory, and nest site scales, and that increased shrub cover also predicted both abundance and territory success. Willow flycatcher nesting success is strongly associated with standing or flowing water or heavily saturated soils (summarized in Green et al. 2003). Meadow size seems to be an important factor for willow flycatcher use. Willow flycatchers in the Sierra Nevada use meadows ranging from 1 to 719 acres (Serena 1982, Harris et al. 1987, Harris et al. 1988, Bombay 1999, Green et al. 2003); but the majority of occupied meadows are larger then 50 acres (Loffland et al. 2014). Willow flycatchers have also been found in riparian habitats of various types and sizes ranging from small lakes or ponds surrounded by willows with a fringe of meadow or grassland, to willow lined streams, grasslands, or boggy areas.

Male willow flycatchers are territorial during the breeding season. Studies in the Tahoe National Forest have found that territory sizes average 0.84 acres (Sanders and Flett 1989). Females may forage outside or at the fringe of the territories defended by males. In addition, after the young fledge the family groups use areas outside of the territories for feeding and cover (M. Flett, pers. comm.). The breeding season begins in late May to early June (Garratt and Dunn 1981) with adults and fledglings generally staying in the breeding areas through August.

Nests are open cupped, usually 3.7 to 8.3 feet above the ground and mostly near the edge of deciduous, riparian shrub clumps (Sanders and Flett 1989, Valentine et al. 1988, Harris 1991). Willow is the most common nest substrate used for breeding in the Sierra Nevada. In the Tahoe National Forest, Sanders and Flett (1989) reported that all 20 nests found at two meadows in 1986 and 1987 were located in willows. More recent studies (between 1997 and 2001) in the north-central Sierra Nevada located 250 nests at 15 meadows, of which only 3 nests were in shrubs other than willow: two in alder, and one in dogwood (Morrison et al. 2000, Bombay et al. 2001). Recently, McCreedy and Heath (2004) located willow flycatchers in the rewatered Rush Creek at Mono Lake, where all nine nests found were in wild rose, even though apparently suitable willows were available. The latter is more likely atypical as a result of extreme management activities rather than being typical under “natural” conditions.

Willow flycatchers forage by either aerially gleaning insects from trees, shrubs, and herbaceous vegetation, or they hawk larger insects by waiting on exposed forage perches and capturing them in flight (Ettinger and King 1980, Sanders and Flett 1989). Sanders and Flett (1989) found that in Perazzo Meadows, willow flycatchers usually flew less than 3.3 feet from a perch when hawking insects, but occasionally flew as far as 33 feet. The selection of nest sites near water appears to be related to increased densities of aerial insects. Willow flycatchers feed upon a wide variety of insect and other arthropod prey (Green et al. 2003). They seem to prefer hymenopterids (bees, wasps, and sawflies), dipterids (deer flies and bee flies), moths and butterflies, and small flying beetles. In the Sierra Nevada, hymenopterids and deer flies are particularly important to their diet (as summarized in Green et al. 2003*)*. Many studies have found a correlation between insect abundance and hydrologic conditions and riparian vegetation condition (as summarized in Green et al. 2003). Some studies have documented that insect abundance has been linked to bird abundance (as summarized in Green et al. 2003).

Potential predators of willow flycatcher nests include a variety of mammalian and avian species (Cain et al. 2003), the occurrence of which varies according to environmental characteristics in different portions of meadows (Cain et al. 2006). In the north-central Sierra Nevada, Cain et al. (2003) reported that nest predation was the primary cause of all willow flycatcher nest failures (76%; 22 of 29 nest failures) during their study from 1999-2000; 41% of nests were successful (fledged at least one young; 20 of 49 nests), 45% were depredated (22 of 49 nests), 8% were abandoned (4 of 49 nests), 4% failed due to weather (2 of 49 nests), and 2% were parasitized (1 of 49 nests). Cain et al. (2003) also documented predator species via trip-wired cameras on simulated willow flycatcher nests (zebra finch eggs placed in vacant yellow warbler nests similar to willow flycatcher nests), finding that the photographed predators in order of decreasing prevalence were various chipmunk species (6 nests), Douglas squirrels (4 nests), short-tailed weasels (3 nests), and a deer mouse (1 nest). Douglas squirrel and chipmunk activity decreased substantially as mean meadow wetness increased, and Douglas squirrel activity was negatively correlated with meadow size (Cain et al. 2003). Cain et al. (2003) found the activity indices of the following potential predators or brood parasites negatively associated with willow flycatcher nest success: Douglas squirrels, short-tailed weasels, Clark’s nutcrackers, Cooper’s hawks, and brown-headed cowbirds. Cain et al. (2006) found Douglas squirrels and chipmunks associated with characteristics common along edges of meadows, short-tailed weasels associated with willows (presumably for protective cover), and mice and long-tailed weasels in a variety of environmental conditions and possibly throughout the meadows. The models they tested suggested that short-tailed weasels may be important predators of the willow flycatcher (Cain et al. 2006).

While brown-headed cowbird parasitism may have been a major contributor to the decline of lowland populations of willow flycatcher, there is less evidence for this in the higher elevations of the Sierra Nevada (Sanders and Flett 1989). In sites monitored in the demography study in the central and northern Sierra Nevada from 1997-2008, mean annual cowbird parasitism rates ranged from 6.8% to 18.4%, depending on the study region and the method of estimating parasitism rates (Mathewson et al. 2009). In the central study area which includes sites in the Tahoe National Forest, the maximum mean annual parasitism rate is calculated at 11% (Mathewson et al. 2009).

Loffland et al. (2014) associated occupied sites with a northward latitude habitat contraction. The demography study reported that for the area studied on the Tahoe National Forest an annual population increase of 6% is needed to stabablize the population and recommended restoring degraded meadows as a tool needed to aid in population recovery (Loffland et al. 2014).

This spatial area used to analyze effects to this species is identified as the location of project proposals buffered by 0.25 miles. This area is sufficient in identifying any potential loss of habitat or disturbance to nesting individuals. The project area does not occur within or near any willow flycatcher habitat.

B. Willow Flycatcher: Effects of the Proposed Action and Alternatives including Project Design Standards

Due to the lack of suitable habitat, this project would not have any direct, indirect, or cumulative effects to this species.

C. Willow Flycatcher: Conclusion and Determination

It is my determination that this project will not affect the willow flycatcher.

GREATER SANDHILL CRANE

Status: USFS R5 Sensitive

A. Greater Sandhill Crane: Existing Environment

The greater sandhill crane is a California State Threatened species and is listed as Sensitive on the Region 5 Forester's Sensitive Species List (USDA Forest Service 1998). The Tahoe National Forest LRMP, as amended, does not provide specific management guidelines for this species. However, standards and guidelines designed to meet Riparian Conservation Objectives apply to the crane and its habitat (listed under the California red-legged frog).

Greater sandhill cranes of the west coast are not hunted, and are protected by the federal Migratory Bird Treaty Act of 1918 (USDA 1994). The California Central Valley population of sandhill cranes is the most western of five distinct populations. A total of 276 cranes were recorded within the state during a breeding pair survey in 1988 (California Department of Fish and Game 1997). In California, greater sandhill cranes winter primarily throughout the Sacramento, San Joaquin, and Imperial Valleys (Grinnell and Miller 1944). Current known breeding populations are located within Lassen, Modoc, Plumas, Shasta, Sierra, and Siskiyou Counties (James 1977, Littlefield 1982, California Department of Fish and Game 1994). In the Tahoe National Forest, a breeding population of approximately 11 pair occur within Carman Valley and Kyburz Flats on the Sierraville Ranger District (Youngblood 1998, personal communication).

California pairs of sandhill cranes generally nest in wet meadow, shallow lacustrine, and fresh emergent wetland habitat, with nests constructed of large mounds of water plants over shallow water (Zeiner et al. 1990, California Department of Fish and Game 1994). Studies in California during 1988 showed water depths averaging 2.3 inches (California Department of Fish and Game 1994). Open meadow habitats are also used (Littlefield 1989). On dry sites, nests are scooped-out depressions lined with grasses (Zeiner et al. 1990). Nesting territory size depends on the quality of available habitat. Little is known about territory size for breeding adults in California.

Cranes do not breed until their fourth year, but then usually mate for life (Johnsgard 1975). Nesting activities begin with courtship in April. Peak breeding occurs in May through July, with nesting usually completed by late August. The average clutch size is two, ranging from one to three. Incubation takes approximately 30 days. Shortly after the second egg hatches, adults lead the young from the nest site and begin feeding them. Each adult generally feeds one chick. Chicks are aggressive toward each other, and, shortly after hatching one becomes dominant. Often this dominance leads one chick to be pushed away from the adults. This may cause the chick to starve or be consumed by a predator (Zeiner et al. 1990, California Department of Fish and Game 1994). Young fledge at about 70 days, but remain with their parents for up to one year (Harrison 1978).

Within nesting territories, water and foraging areas are the primary habitat elements necessary for reproductive success. Young cranes (colts) depend mostly on invertebrates during their first five or six weeks and sometimes starve to death when invertebrates decrease with water levels (Pacific Flyway Council 1997). In dry years colts are moved upland, where they feed primarily on grasshoppers and other insects (California Department of Fish and Game 1994). Adults feed on grasses, forbes, cereal crops, roots, tubers. Animal matter such as insects, mice, crayfish and frogs, is taken opportunistically, but should not be considered a major component of their diet.

Recruitment of young is suppressed by predation at most breeding areas within their range (California Department of Fish and Game 1994). Predation from coyotes, common raven and raccoons are a major factor in low nesting success, especially in years of low precipitation (Littlefield 1989, California Department of Fish and Game 1994, Pacific Flyway Council 1997). Preliminary studies indicate that up to 45% of egg losses and up to 76% of young crane mortality may be attributed to predation (Ivey 1995). Surveys conducted by R. Schlorff on greater sandhill cranes in California between 1979 and 1986 averaged a 5.6% recruitment rate, although a recruitment rate of 12% is necessary for population stability (Miller et al. 1972).

Spring and summer livestock grazing may cause a loss of nests and young due to nest desertion and trampling of young (Littlefield 1989). This can be extremely detrimental to breeding cranes, especially if water is limited (California Department of Fish and Game 1994). Lowering of ground water tables often results in stream down cutting with subsequent drying and degradation of wetland habitats (California Department of Fish and Game 1994).

The project proposals, buffered by 0.25 miles is the spatial area used to analyze effects to this species. This area would sufficienty address any potential loss of habitat or any noise disturbances that could occur to individuals from project implementation or trail use over the long term. There is no suitable habitat within the project area.

B. Greater Sandhill Crane: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct, Indirect, and Cumulative

Due to the lack of suitable habitat for this species, there are no direct, indirect, or cumulative effects that would occur.

C. Greater Sandhill Crane: Conclusion and Determination

It is my determination that implementation of this project will not affect the greater sandhill crane.

FOREST CARNIVORES – GENERAL DISCUSSION

The marten and wolverine are carnivores found in the Sierra Nevada of California and are collectively known as "forest carnivores." They are listed as Sensitive on the Region 5 Forester's Sensitive Species List. At the present time, forest carnivore survey protocols can only determine presence. If no individuals are detected, the absence of individuals cannot be concluded with any level of confidence. Until recently, most sighting information on forest carnivores in the Sierra Nevada has been incidental and usually unverified.

The Tahoe National Forest completed a forest wide habitat management plan for these species in 1992, the Recommendations for Managing Late Seral Stage Forest and Riparian Habitats on the Tahoe National Forest (Chapel et al. 1992). This was the basis for inclusion of an old forest/riparian corridor system for each project analyzed prior to October 1, 1998. On May 1, 1998, the Regional Forester requested that each forest complete a mesocarnivore network. The Tahoe National Forest completed that network on September 29, 1998, incorporating new information that focused the network on the needs of marten and fisher instead of the more wide-ranging old-forest/riparian concept. Maps and information on how the network was developed are on file at the Supervisor's Office. This information, in conjunction with the land allocations identified in the Sierra Nevada Forest Plan Amendment (USDA Forest Service 2001), is used during site-specific project analysis to identify potential movement corridors between habitat patches and when developing desired conditions.

PACIFIC MARTEN

Status: USFS R5 Sensitive

A. American Marten: Existing Environment

This species was previously classified as American marten (*Martes americana*) but recent genetic and morphological evidence have led to a re-classification as Pacific marten (*Martes caurina*) and of the subspecies *sierrae* (Aubry et al. 2012).

The marten historically occurred in forests throughout North America, experienced reductions in portions of its range due to intensive trapping and reduction in habitat quality, and has since been reestablished in portions of the historic range with natural expansions and with the aid of translocations (summarized in Kucera et al. 1995). In California, marten were trapped for fur until prohibited in 1946 in the extreme northwestern portion of the state, and throughout the State in 1953 (summarized in Kucera et al. 1995). The Humboldt marten subspecies was thought to be possibly extinct from the coastal range at one time (refer to Kucera et al. 1995, and Zielinski and Golightly 1996 as cited in Zielinski et al. 2001), but marten are now known to exist in a small area in California's north coast range (Zielinski et al. 2001, Slauson 2003, Slauson et al. 2007). Marten appear to be well distributed within their geographic ranges, including in the Sierra Nevada (Zielinski et al. 2001); however, Zielinski et al. (2005) reported a gap in the current distribution centered on Plumas County which was not historically present.

The largest potential threat to marten and their habitat resulting from forest management is the effect of forest thinning since clear-cutting is virtually non-existent on public lands. Harvesting on private land may still have significant effects to marten habitat availability. The effects of forest fragmentation has been reported in numerous cases in the literature, mainly describing the sensitivity of martens to the effects of forest fragmentation (Bissonette et al. 1997, Chapin et al. 1998, Hargis et al. 1999, Potvin et al. 2000 in Zielinski 2013), but in these cases, the fragmentation is typically due to regeneration or clear-cut harvests. A recent study (2010-2013) on the Lassen National Forest concluded that marten avoid openings and select for home ranges with fewer patches with simple forest structures (Moriarty 2014). Fuller and Harrison (2005) evaluated the effects of partial harvests on martens in Maine. They found martens used the partial harvest stands primarily during the summer, where 52-59 percent of the basal area was removed in partial harvests. Marten home ranges were larger when they used partial harvest stands, indicating poorer habitat quality in these areas. Partial harvested areas were avoided during the winter, presumably because they provided less overhead cover and protection from predators. How this study relates to predicting the effects of thinning in marten habitat in the Sierra Nevada is unclear, but it may suggest that martens would likely be associated with the more dense residual areas in thinned units and may also increase their home ranges, which may lead to decreased population density. The negative effects of thinning likely result from reducing overhead cover. Thinning from below, which retains overstory cover, probably has the least impact on marten habitat, provided they retain sufficient ground cover. Downed woody debris provides important foraging habitat for martens. Andruskiw et al. (2008) found that physical complexity on or near the forest floor, which is typically provided by coarse woody debris, is directly related to predation success for martens; when this complexity is reduced by timber harvest (a combination of clear-cut and selection harvests with subsequent site preparation in their study area), predation success declines. Marten home ranges in uncut forests had 30 percent more coarse woody debris (> 10 cm diameter) from all decay classes combined than in cut forests (Andruskiw et al. 2008).

In the northern Sierra Nevada marten generally occur at elevations of 3,400 feet to 10,400 feet, averaging 6,600 feet. Kucera et al. (1995) describe the distribution of the marten in California from eastern Siskiyou and northwestern Shasta Counties through the western slope of the Sierra Nevada to northern Kern County, and on the eastern slope of the Sierra Nevada as far south as central-western Inyo County. In the southern Cascades and northern Sierra Nevada, Kirk (2007) noted that 85% of contemporary marten detections in his analysis occurred above 6,000 feet elevation (despite a reduced survey effort at these higher elevations), 15% of detections were between 3,000 and 6,000 feet, and no detections of marten occurred below 3,000 feet elevation. They most often occur at somewhat higher elevations than the fisher (Freel 1991, Zielinski et al. 2005, Zielinski 2013). In contrast to the fisher, the marten distribution in California corresponds closely to the regions of the heaviest snowfall in the southern Cascades and the Sierra Nevada

(Krohn et al. 1997). This may be due to lower mobility of the fisher in soft snow, as the foot-loading (ratio of body mass to total foot area) of the fisher is >2 times higher than the marten (Krohn et al. 2005). Where their ranges overlap, there may be negative competitive interactions between the fisher and marten (Krohn et al. 1995, Krohn et al. 1997). Fuller and Harrison (2005) note that fishers are a principal arboreal predator of martens in Maine.

Preferred forest types in the Sierra Nevada include mature mesic forests of red fir, red fir/white fir mix, lodgepole pine, subalpine conifer, and Sierran mixed conifer (Freel 1991). CWHR types 4M, 4D, 5M, 5D, and 6 are moderate to highly important for the marten (USDA Forest Service 2001). Analysis of effects to marten weighs heavily on the preferred habitat types, but consideration is given for the utilization of other marginal habitat types. Forest stands dominated by Jeffrey pine do not appear to support marten in the Tahoe National Forest (Martin 1987), as evidenced by the lack of marten detections in pure eastside pine (some of which were adjacent to mixed conifer stands which did contain marten detections) during systematic surveys conducted on the eastside of the Tahoe National Forest (data on file at Sierraville Ranger District).

Preferred habitat is generally characterized by dense canopy, multi-storied, multi-species late seral coniferous forests with a high number of large (> 24 inch dbh) snags and downed logs (Freel 1991). Late- and old-structure forests (with larger diameter trees and snags, denser canopy and more canopy layers, and plentiful course woody material) are thought to provide ample rest and den sites, protection from avian and mammalian predators, and foraging sites (Bull et al. 2005). Data from some studies shows that use of habitat by marten does not necessarily rely on high levels of canopy cover, but likely involves a complex interaction of habitat variables, at both small and large scales, which provide for their life history requirements and minimizes the risk of predation (refer to Soutiere 1979, Drew 1995, Chapin et al. 1997, and Slauson 2003). Koehler and Hornocker (1977) suggested that while open meadows and burns may be avoided by marten in winter when they are under a heavy snowpack, these areas may be used in the summer, or in low snow years, if they provide adequate cover and food.

Marten have been found to be generally associated with moist conifer-dominated forest conditions (eg. Spencer et al. 1983, Martin 1987, Buskirk et al. 1989, Wilbert et al. 2000, Mowat 2006, Baldwin and Bender 2008). Studies in the Sierra Nevada indicate martens have a strong preference for forest-meadow edges, and riparian forest corridors used as travel ways appear to be important for foraging (Spencer et al. 1983, Martin 1987). Spencer et al. (1983) found that in the lower Sagehen Creek basin on the eastside of the Tahoe National Forest below approximately 6,700 feet elevation, marten strongly preferred riparian lodgepole pine habitat and selected against brush, mixed conifer, and Jeffrey pine habitats; riparian areas were used more for activity than resting, and mixed conifers were used more for resting than activity. In the upper Sagehen basin above approximately 6,700 feet elevation, marten were found to strongly prefer red fir habitat associations for both resting and activity (Spencer et al. 1983). Spencer et al. (1983) found that marten preferred forest stands with 40-60% canopy cover at both resting and foraging sites and avoided stands with less than 30% canopy cover.

Coarse woody debris is an important component of marten habitat, especially in winter, by providing structure that intercepts snowfall and creates subnivean tunnels, interstitial spaces, and access holes (Andruskiw et al. 2008). Marten rest sites in winter are most often in subnivean sites, most often associated with course woody debris, especially during periods of colder temperatures and recent precipitation, but can also be found in association with rocks (Buskirk et al. 1989, Bull and Heater 2000, Wilbert et al. 2000). Rest sites next to course woody debris in the subnivean space offer thermal insulation in colder temperatures (Buskirk et al. 1989). Zielinski et al. (1983) suggested that marten activity varied to allow them to take advantage of subnivean dens utilized by their prey. Sherburne and Bissonette (1993) found marten more likely to utilize subnivean access points in areas that contained more abundant prey. They also found that when coarse woody debris covered a greater percent of the ground, marten use also increased (Sherburne and Bissonette 1993). Older growth forests with accumulated coarse woody debris provide the forest floor structure necessary to enable marten to forage effectively during the winter (Sherburne and Bissonette 1993). In Ontario, Andruskiw et al. (2008) found that despite having lower levels of course woody debris, the availability of subnivean access points was not less in regenerating forest compared to uncut forest due to access points created by low-reaching branches of young conifer trees; however, only the subnivean access points created by course woody debris contained small mammals and were used by marten.

Marten home ranges are large by mammalian standards, particularly for their size (Buskirk and Ruggiero 1994, Buskirk and Zielinski 1997). Martens exhibit a high level of variation in home range size throughout their range, and generally exhibit a low level of same-sex overlap (Bull and Heater 2001). From numerous studies across the range of the marten, Powell (1994) calculated the mean home range size for males to be 2,000 acres and for females 570 acres (as cited in Powell et al. 2003). Marten home range sizes in the Sierra Nevada have been reported to vary from approximately 420 to 1,800 acres for males, and 170 to 1,400 acres for females (summarized data from Simon 1980, Spencer 1981, Martin 1987, and Zielinski et al. 1997 as reported in Buskirk and Zielinski 1997). Variation in home range size may be a function of prey abundance or habitat quality (Ruggiero and Buskirk 1994). In northeastern Oregon, Bull and Heater (2001) found that home range size was not correlated to the amount of unharvested forest in their study. In Maine, Chapin et al. (1998) found that regenerating forest (stands harvested in approximately the past 15 to 20 years) composed a median of 22% (range 9-40%) of male home ranges and 20% (range 7-31%) of female home ranges; the largest residual forest patch (contiguous areas composing adjacent stands of mid- to late-successional forest) in the home range composed a median of 75% (range 30-90%) of male home ranges and 80% (range 51%-93%) of female home ranges.

Drew (1995) suggested that some fine-scale selection factor not linked to foraging strategy, such as minimizing the risk of predation by avoidance of open areas, appears to influence habitat selection, and recommended maintenance of landscape connectivity to prevent isolation of forest patches. Kirk (2007) found the best association for marten occurrence at the largest scale he modeled (30.9 mi^2), with amount of habitat, number of habitat patches, and land ownership category emerging as important variables, suggesting selection based upon broad scale landscape conditions. The size of openings that martens will cross in the Sierra Nevada or Cascades is currently under study (Zielinski 2013). However, in the Rocky Mountains, the average width of clear cuts (openings) crossed by martens was 460 feet; this distance is significantly less than the average width of clear cut openings that martens encountered but did not cross (average = 1,050 feet) (Heinemeyer 2002). Moreover, martens were more likely to cross larger openings (max distance = 600 feet.) that had some structures in them (i.e., isolated trees, snags, logs) than smaller openings (average distance = 160 feet) that had no structures (Heinemeyer 2002). Cushman et al. (2011) reported that snow-tracked martens in Wyoming strongly avoided openings and did not venture more than 55 feet from a forest edge.

In the Sagehen Creek basin in Tahoe National Forest, Moriarty et al. (2011) found that marten detections decreased from an average detection rate of 65% in the early 1980s (Spencer 1981, Zielinski 1981, Martin 1987) to 4% in her study conducted from 2007-2008, based on similar but not identical methodology. Analysis of prior research in this area showed that the distribution of marten detections changed spatially from a semi-uniform distribution in the upper and lower basin in 1980s to detections that were clustered in the southwest corner of the upper basin by the early 1990s (Moriarty et al. 2011). The reasons for the apparent decrease in marten abundance were not clear, but may have included reduction of habitat quality, increase in habitat fragmentation, loss of important microhabitat features such as snags and down woody material, or other factors (Moriarty et al. 2011). From 1984-1990 more than 30% of the forested habitat in the Sagehen basin was impacted by various logging treatments (Moriarty et al. 2011). Moriarty et al. (2011) suggested that rather than amount of habitat (which did not change significantly), it is likely that the size of patch core areas, distance between patches, spatial configuration of patches, and microhabitat features within patches may be more important for marten persistence.

In Yosemite National Park, Hargis and McCullough (1984) found that marten will cross meadows less than or equal to 50 meters wide in winter with no cover, and use scattered trees for cover across meadows greater than 50 meters wide to a maximum of 135 meters. Marten traveled in all major habitat types, without any detectable habitat preferences, but did not pause in openings (only in forests, ecotones, and on frozen streams); locations where they paused were associated with closer distance to the nearest tree, percentage of overhead cover, and height (< 3 meters) of overhead cover (Hargis and McCullough 1984). Larger open areas which lack ground cover may pose a predation risk for the marten (Drew 1995). Drew (1995) found that habitat dominated by defoliated stems (due to tree mortality from bug infestations in his study) may provide sufficient cover.

Prey species abundance is a critical component of the habitat and there is some dietary overlap with the fisher, particularly in the southern Sierra Nevada where they occur sympatrically (Zielinski and Duncan 2004). Both species prey heavily upon squirrels, but marten diet has been found to be diverse, including a variety of mammals, birds, reptiles, fish, insects, seeds, and fruits (Koehler and Hornocker 1977, Soutiere 1979, Hargis and McCullough 1984, Zielinski and Duncan 2004). Marten prey items vary seasonally and appears to depend on availability. Simon (1980) found insects dominating the diet in summer and fall, while Douglas squirrels (*Tamiasciurus douglasii*) provided the bulk of winter and spring nourishment. At Sagehen Creek, CA, within the Truckee Ranger District, Zielinski (1983) found microtine rodents the most frequent year-round prey. Douglas squirrels, snowshoe hare, northern flying squirrel, and deer mouse were taken almost exclusively during the winter; and squirrels and chipmunks formed the largest component of the diet from late spring through fall.

Numerous and heavily traveled roads are thought to be undesirable in order to avoid habitat disruption and/or animal mortality. Roads may decrease prey and food availability for marten as well as fisher (Allen 1987; Robitaille and Aubry 2000) due to prey population decreases resulting from road kills and/or behavioral barriers to movement. Other studies have shown that occasional one and two lane forest roads should not limit marten movements (Chapin et al. 1997; Mowat 2006; Kirk 2007). In two study sites in California (Lake Tahoe Basin Management unit and Sierra National Forest), Zielinski et al. 2008 found that off-highway vehicle and over-the-snow vehicle use (at least up to 1 vehicle per 2-hour time period) had no effect on marten occurrence, circadian activity, or sex ratio. In west-central Alberta, Webb and Boyce (2009) found that no traplines with consistent marten harvests through time had >36% of the trapline developed; in their study roads and oil and gas wells were the primary form of development.

Marten sightings within the Tahoe National Forest generally follow a band encompassing the higher elevations on either side of the Pacific Crest. Winter surveys for forest carnivores have confirmed marten presence within the Tahoe National Forest, generally spanning the Pacific Crest to the northeast and east of this project.

The trail proposals were buffered by 0.25 miles to account for potential direct effects from trail construction or decommissioning within marten habitat, and the potential effects to disturbing individuals extending into adjacent marten habitat. Marten are known to occur within the headwaters of the North Yuba River and the Lakes Basin area of the District east and west of the Gold Lake Highway. Their preferred habitat is where lodgepole pine interfaces with meadow habitat. None of the project area occur within suitable habitat for this species.

B. American Marten: Effects of the Proposed Action and Alternatives including Project Design Standards

Management Requirements that reduce effects to this species include the following:

1. Where practicable, locate trail to avoid the need to fall large trees and snags, or those displaying wildlife use (cavities, nests). Fall and leave hazardous snags to recruit dead wood.

2. If new Threatened, Endangered, or Forest Service Sensitive (TES) species are listed or discovered, or nesting TES are found within 0.25 mile of activities, a limited operating period will be implemented as recommended by a qualified biologist.

Direct and Indirect Effects

Individuals could be disturbed during foraging, denning, or resting, if they are present during project implementation. Although late successional, closed canopy forested habitat is present within this project area, preferred habitats of high elevation meadow habitat interfacing with lodgepole pine forests and late successional forests is not present in the project habitat. Therefore, this species is not likely to commonly occur in the project area, den, or be present during project implementation.

Cumulative Effects

The cumulative effects area includes all of the trail proposals, buffered by 0.25 miles. This area would account for potential effects of directly removing habitat or effects from noise disturbances that could occur to individuals from trail construction, relocation, decommissioning, or in the future from use of the trails over the long term. There is no private land that overlaps with this project area. Cumulative effects within the project area occurs primarily from noise disturbances related to recreational use along of the trails, dispersed recreational use off of roads and trails, and road and trail maintenance to manage encroaching vegetation and fell hazard trees. Because the majority of the project lies within the East-West Yuba RARE II area, there is little additional vegetation management that is conducted. There are no additional projects that are planned within the project area. Because this project includes both trail decommissioning and trail re-location, and new trail connections occur outside of suitable habitat, there is little difference in any additional effects that might occur from this project

C. American Marten: Conclusion and Determination

It is my determination that implementation of this project may affect individuals, but is not likely to result in a trend toward Federal listing or loss of viability for the American marten within the planning area of the Tahoe National Forest. In the absence of a range wide viability assessment, this viability determination is based on local knowledge of the American marten as discussed previously in this evaluation, and professional judgment.

WOLVERINE

Status: USFS R5 Sensitive

A. Wolverine: Existing Environment

The wolverine is currently proposed for listing by the US Fish and Wildlife Service and a new status review has been initiated (2016) to determine if the northern distinct population segment warents listing. The wolverine is also considered a California State Threatened species, and is listed on the USFS Regional Forester's Sensitive Species list. The wolverine was petitioned for listing as threatened or endangered under the Endangered Species Act in 2008 but upon status review in 2008 the USFWS determined it was not warranted listing (USFWS 2008; 73 FR 12929).

Wolverines are distributed across the circumpolar northern hemisphere (Banci 1994; USFWS 2011) and their range extends south along connected mountain ranges (Pasitschniak-Arts and Larivière 1995). Historically in North America, they were found from Alaska to eastern Canada (Banci 1994; Pasitschniak-Arts and Larivière 1995) and south across the Canada-US border provinces and states, and extended south along the Rocky Mountains to Arizona and New Mexico and along the west coast mountains and Sierra Nevada Mountains (Banci 1994; Pasitschniak-Arts and Larivière 1995). The southern extent of the wolverine's range is discontinuous and is found in mountainous terrain. In the contiguous US, wolverines are found in, Montana, Wyoming, Idaho, Oregon, Washington, and, recently, California (Moriarity et al. 2009; USFWS 2011) and Colorado (USFWS 2010).

Since the last historic specimen was collected in California in 1922 (Fry 1923, and Grinnell et al. 1937 as cited in Aubry et al. 2007), there have been periodic anecdotal sightings (lacking conclusive physical evidence) of the wolverine in California including many in the Tahoe National Forest, though these anecdotal sightings should be interpreted with caution (refer to McKelvey et al. 2008). Verifiable records were practically absent since the early twentieth century (Moriarity et al. 2009) until a male wolverine was photographed at baited camera stations on the Tahoe NF and adjacent Sierra Pacific Industries land in 2008 through 2015 (Moriarity et al. 2009; USFWS 2010; USFS NRIS records database 2015, CDFW 2014). These records are north of Interstate 80 in Nevada and Sierra counties, west and south of Sierraville, California. The USFWS considers the Sierra Nevada Mountains to be part of the wolverine's current range, but a population has not been reestablished (the single male identified in 2008 does not make a population) (USFWS 2010). After a status review in 2010 the USFWS concluded that although there is one verified wolverine in the Sierra Nevada and there may be others dispersing into the area, there is no evidence that California currently has a wolverine population.

The historic geographical range of the wolverine in California was originally described from the vicinity of Mount Shasta to Monache Meadows in Tulare County (Grinnell 1913 as cited in Shempf and White 1977), which was later refined to Lake Tahoe through Tulare County in the central and southern Sierra Nevada (Grinnell 1933 and Grinnell et al. 1937, as cited in Shempf and White 1977). Fry (1923) noted that the wolverine was found in the high Sierra between 6,500 and 13,000 feet, that it was becoming very rare with individuals few and scattered where still found, and were most abundant in the vicinity of Mt. Whitney and Sequoia National Park. Grinnell et al. (1937) noted that "the wolverine in California is found chiefly in the Boreal life zone...at the time of heavy snowstorms in midwinter, wolverines have been found as low as 5,000 feet on the westslope of the Sierra Nevada...But ordinarily the wolverine is not known to come below 8,000 feet, even in the severest storms of winter." Grinnell et al. (1937) estimated that there were no more than 15 pairs of wolverines left in California in the early 1930s. Based on reported sightings from various sources, Schempf and White (1977) described the wolverine distribution in California as a broad arc from Del Norte and Trinity counties (Yocum 1973) east through Siskiyou and Shasta Counties (Wildlife Management Institute 1974 as cited in Schempf and White 1977), and south through the Sierra Nevada to Tulare County (Jones 1950 and 1955). Kovach (1981) expanded this range to include the White Mountains in Mono County. Shempf and White (1977) described the elevational range in the North Coast mountains from 1,600 to 4,800 ft, in the northern Sierra Nevada from 4,300 to 7,300 ft, and in the southern Sierra Nevada from 6,400 to 10,800 ft. However, Aubry et al. (2007) scrutinized the historical records, and suggest that historically the wolverine population in the Pacific states was disjunct, with a large gap in distribution from the populations in the north Cascades and the Rocky Mountains to the historic California wolverine population, which only occurred in the central and southern Sierra Nevada. This conclusion is reinforced by genetic analysis conducted by Schwartz et al. (2007). Schwartz et al. (2007) compared DNA from seven historical specimens of wolverines from California with those from other locations throughout their holarctic distribution; their results indicate that wolverines from California likely genetically diverged over 2,000 years (and possibly much longer) ago from any other population, and are genetically more similar to Eurasian wolverines than to other North American populations.

Wolverines in the southern part of the Pacific Northwest Coast and Mountains ecoprovince are becoming isolated from the northern portion of the ecoprovince by heavy development in British Columbia. However, occasional reports within the Thomson-Okanogan Highlands ecoprovince of British Columbia and Washington suggest that this may be a dispersal corridor. It is also possible that wolverines have become isolated within the Sierra Nevada ecoprovince of California because of human activities. (Banci 1994).

There are probably 250 to 300 wolverines in the contiguous US and these are distributed among five subpopulations: Northern Cascades (WA), Idaho, the Greater Yellowstone Ecosystem, northern Montana, and the Crazy and Belt population (west-central MT) (USFWS 2011). Furthermore, the distance between these subpopulations is far enough such that dispersal, and therefore genetic exchange, is infrequent (USFWS 2011). The USFWS (2011) indicates at least 400 breeding pairs would be necessary to sustain genetic variability over the long term in the contiguous US, absent immigration. Genetic structure in the southern and eastern extremes of the distribution of wolverines is relatively high, possibly reflecting the fragmented nature of these populations at the periphery of their historical range (Kyle 2002).

There is one known wolverine in California. The two source population centers (the North Cascades and western Rocky Mountains) are the nearest to the area, and the Idaho population is the closest. However, the Idaho population is still at least 400 miles away, which makes for a long dispersal distance. Hair and scat samples were acquired and it was genetically determined that this animal most likely from the western Rocky Mountains, and perhaps more accurately, central Idaho (Moriarity et al. 2009).

In the contiguous US, wolverine populations appear to have increased during the last 50 years, as indicated by recolonization of areas from which they were previously extirpated (Aubry et al. 2007). Recently, dispersing males have been documented in California (Moriarty et al. 2009) and Colorado (Inman et al. 2010). These are the first verifiable wolverine records in these states since 1924 and 1919, respectively (Aubry et al. 2007), and are likely associated with current population expansions such as the recent observed recolonization of the northern Cascade Range in Washington (K. Aubry, unpublished data).

Males reach sexual maturity at two and a half years; females may breed at age one (Banci 1994). Mating occurs during the summer (May through August), followed by delayed implantation (Pasitschniak-Arts and Larivière 1995). Because implantation can occur from November to as late as March, kits can be born as early as January or as late as April (Banci 1994), but are usually born in February or March (Pasitschniak-Arts and Larivière 1995). One to four kits make up the typical litter (Banci 194; Pasitschniak-Arts and Larivière 1995). Kits develop rapidly, are weaned in seven to ten weeks, and are full grown in about seven months (Banci 194; Pasitschniak-Arts and Larivière 1995).

Dispersal by young is different for each sex. Females typically stay near where they were born, whereas males will not. The longest recorded dispersal distance by a male was 378 kilometers (about 230 miles) from Alaska to Yukon (Banci 1994). If the wolverine from California was a dispersing male from Idaho, it traveled about 400 miles, and upwards of 600 miles if from other subpopulation centers.

Broadly, wolverines are restricted to boreal forests, tundra, and western mountains. The vegetation zones occupied by wolverines include the Arctic Tundra, Subarctic-Alpine Tundra, Boreal Forest, Northeast Mixed Forest, Redwood Forest, and Coniferous Forest (Banci 1994). Researchers have generally agreed that wolverine "habitat is probably best defined in terms of adequate year-round food supplies in large, sparsely inhabited wilderness areas, rather than in terms of particular types of topography or plant associations" (Kelsall 1981).

Wolverines naturally occur at low densities, and require cold areas that maintain deep, persistent snow cover into the warm season for successful denning, (USFWS 2011). Within the contiguous United States wolverine habitat is restricted to high-elevation areas in the West (Figure 2). Their current distribution includes functioning populations in the North Cascades Mountains and the northern Rocky Mountains, as well as populations that have not yet reestablished in the southern Rocky Mountains and the Sierra Nevada (Ibid).

Consistent with field observations indicating that all wolverine reproductive dens are located in areas that retain snow in the spring (Magoun and Copeland 1998), Aubry et al. (2007) concluded that the distribution of persistent spring snow cover was congruent with the wolverine's historical distribution in the contiguous United States. This relationship was further supported by the findings of Schwartz et al. (2007) that showed historical wolverine populations in the southern Sierra Nevada of California, which occupied a geographically isolated area of persistent spring snow cover, were genetically isolated from northern populations.

Recently one researcher compiled (Inman 2012) most of the extant spatial data on wolverine denning and habitat use to test the hypotheses that wolverines require snow cover for reproductive dens (Magoun 1998), and that their geographic range is limited to areas with persistent spring snow cover (Aubry et al. 2007). Although Aubry et al. (2007) analysis covered only North America and used relatively coarse EASEGrid Weekly Snow Cover and Sea Ice Extent data (Inman 2012), these relationships with finer scale snow data (0.5-km pixels) obtained throughout the Northern Hemisphere and were confirmed from the Moderate-Resolution Imaging Spectroradiometer (MODIS) instrument on the Terra satellite (Ibid). Specifically, the data was compiled and evaluated the locations of 562 reproductive dens in North America and Scandinavia in relation to spring snow (Ibid). All dens were located in snow and 97.9% were in areas identified as being persistently snow covered through the end of the wolverine's reproductive denning period (15 May; Aubry et al. 2007) based on MODIS imagery. Additionally, it was that areas characterized by persistent spring snow cover contained 89% of all telemetry locations from throughout the year in nine study areas at the southern extent of current wolverine range (Inman 2012). Excluding areas where wolverines were known to have been extirpated recently, persistent spring snow cover provided a good fit to current understandings of the wolverine's circumboreal range (Ibid). Moreover, it was found that the genetic structure of wolverine populations in the Rocky Mountains was consistent with dispersal within areas identified as being snow covered in spring, and strong avoidance of other areas. Thus, the areas with spring snow cover that supported reproduction (Ibid) could also be used to predict year-round habitat use, dispersal pathways, and both historical (Aubry et al. 2007) and current ranges ((Inman 2012). Security cover for kits may also be enhanced during winter; snow tunnels or snow caves are characteristic natal and maternal dens for wolverine in many areas (Banci 1994).

Information on the use of natal dens in which the kits are born by wolverines in North America is biased to tundra regions where dens are easily located and observed. These natal dens typically consist of snow tunnels up to 60 meters in length (Banci 1994). Bedding does not appear necessary, inasmuch as kits were found in shallow pits dug on the ground (Ibid). Snow tunnels in northwest Alaska were also used by lone wolverines (Ibid), suggesting that they dig tunnels or use existing tunnels as resting sites as well. Natal dens above treeline appear to require snow 1-3 meters deep (Banci 1994) that persists into spring. In Finland, one researcher believed that dens that wolverine had dug themselves were preferred, because caves were rarely used, although available (Ibid). Little is known of the distribution of den sites in the landscape. The proximity of rocky areas, such as talus slopes or boulder fields, for use as dens or rendezvous sites was important for wolverines in Norway (Ibid), in the Soviet Union (Ibid), and in Idaho (unpublished data in Copeland 1993). Natal dens may be located near abundant food, such as cached carcasses or live prey (Rausch and Pearson 1972; Banci 1994).

Limited information is available on dens in forested habitat. In northern Lapland, most of the dens in forests were associated with spruce (Picea sp.) trees; five consisted of holes dug under fallen spruces, two were in standing spruces, and one natal den was in-side a decayed, hollow spruce and it was reported that dens in Kamchatka were usually constructed in the "hollows" (cavities) of large trees (Banci 1994). Rarely, kits have been found relatively unprotected, on branches and on the bare ground. If females are disturbed they will move their kits, often to what appear to be unsuitable den sites (Ibid). It has been hypothesized that one of the factors affecting the selection of a natal den site was the ease with which it could be adapted to a den. It was reported dens in abandoned beaver lodges (Banci 1994), old bear dens, creek beds, under fallen logs, under the roots of upturned trees, or among boulders and rock ledges. In Siberia, dens were found in caves, under boulders and tree roots, and in accumulations of woody debris consisting of broken or rotted logs and dry twigs (Ibid). Natal dens in Montana were most commonly associated with snow-covered tree roots, log jams, or rocks and boulders (Ibid).

What is understood about home range size and habitat use is generally from remote, undeveloped northern regions and generally is not known for the Sierra Nevada. Schempf and White (1977) extrapolated from locations of anecdotal reports of wolverines in the northern Sierra Nevada that they use mixed conifer habitat (8 of 16 reports), lodgepole (4 of 16 reports), and fir (3 of 16 reports). White and Barrett (1979) believed that wolverines in California are highly dependent upon mature conifer forests for survival in winter, and generally move downslope in winter into heavier timber where food is available. In their preliminary search for study animals previous to capture for their demographic analysis, Squires et al. (2007) considered all forested areas (excluding ponderosa pine forest) and areas above tree line as potential wolverine habitat.

Wolverines can travel long distances in their daily hunting, 30-40 km being "normal" (Banci 1994). Adult males generally cover greater distances than do adult females (Ibid) and may make longer and more direct movements (Ibid). During late winter, lactating females with young move less than solitary adult females (Ibid). In May and June, hunting mothers periodically return to their young that have been left at rendezvous sites (Ibid). In northwest Alaska, females returned to rendezvous sites at least daily (Ibid). Kits were moved to new rendezvous sites every 1-9 days and more frequently as they grew older (Magoun 1985). By June, kits were moved every 1-2 days (Ibid). When her kits were 4-11 weeks old, a female in central Idaho used 18-20 den sites, moving her kits a total of about 26 km (unpublished data in Copeland 1993).

In northwest Montana, wolverines of both sexes made frequent long movements out of their home ranges that lasted from a few to 30 days, and they always returned to the same area (Hornocker and Hash 1981). These long-distance movements appear to be temporary and not attempts to expand the home range. Whether these movements are exploratory or whether wolverine are returning to previously known feeding locations is unknown (Banci 1994). Except for females providing for kits or males seeking mates, movements of wolverine are generally motivated by food. Wolverines restrict their movements to feed on carrion or other high quality and abundant food sources (Gardner 1985; Banci 1988).

Home ranges of adult wolverine in North America range from less than 100 square kilometers (km2) to over 900 km2. The variation in home range sizes among studies partly may be related to differences in the abundance and distribution of food. (Banci 1994). Young females typically establish residency next to or within the natal home range (Banci 1994). Although some immature females disperse, males are more likely to do so. Male wolverines may disperse either as young-of-the-year or as subadults (Banci 1994; Banci 1988). Dispersal can include extensive exploratory movements (Banci 1994, Banci 1988).

Winter home ranges typically overlap with those used in the snow-free season but also include different habitats, even if there are no significant differences in the size of seasonal home ranges (Banci 1994, Banci 1998). Differences between seasonal home ranges can be attributed to changes in prey distribution and availability. Wolverines of both sexes appear to maintain their home ranges within the same area between years (Banci 1994; Banci 1988). There may be slight changes in the yearly boundaries of home ranges with the addition of juvenile females adjacent to the natal area, with mortality, and with immigration. For example, when a resident dies, a neighbor may assume part of the vacant home range (Banci 1994; Banci 1988).

In central Idaho, Copeland et al. (2007) examined habitat associations by aerial and ground radio-tracking, finding that elevation was the strongest and most consistent explanatory variable across all of their logistic regression models. Excluding mountaintops, wolverines favored higher elevations; 83% of all wolverine use points occurred in a relatively narrow elevation band from approximately 7,200 to 8,500 feet elevation, and there was only a minor but statistically significant seasonal shift downward in elevation (mean of 325 feet) during winter except for adult females which did not exhibit a significant downward shift (Copeland et al. 2007). In the central Idaho study area, alpine scree habitats associated with talus and open mountaintops above timberline generally occurred above approximately 8,850 feet elevation; modeling indicated that these habitats were used but not beyond availability in summer, and were avoided in winter (Copeland et al. 2007). In northwestern Montana, Hornocker and Hash (1981) also found a seasonal trend in elevational use; with mean elevation in winter (4500 feet) lower than those in spring (5500 feet), summer (6300 feet), and fall (6200 feet). In central Idaho, Copeland et al. (2007) found that northerly aspects were generally preferred in both seasons (except for adult males in the summer), possibly related to the increased prevalence of the shrub-grass vegetation type (which the wolverines strongly avoided) on southerly aspects. In contrast, in northwestern Montana Hornocker and Hash (1981) found that while all aspects were used, the easterly and southerly aspects received the majority of consistent use. In the northwestern Montana study area, Hornocker and Hash (1981) found that various types of topography were used; slopes were used 36%, basins 22%, wide river bottoms 14%, and ridge tops 8%. In central Idaho, Copeland et al. (2007) found in their models that steep slopes were a strong variable for wolverine presence in summer, most notably in adults, but noted that this may have been reflective of the preference for the higher elevations in which the steeper slopes occur with more frequency.

Hornocker and Hash (1981) found that large areas of medium or scattered mature timber accounted for 70% of all relocations, and areas of dense young timber were used least. Hornocker and Hash (1981) noted that wolverines were found in wet timber, dry timber, and alpine areas during 23%, 31%, and 16% of their relocations, respectively, and were rarely found in burned or wet meadow areas. Hornocker and Hash (1981) found that wolverines appeared reluctant to cross openings of any size such as recent clear-cuts or burns; they state: "Tracking revealed that wolverines meandered through timber types, hunting and investigating, but made straight-line movements across large openings." They note that edge and ecotonal areas are used (Hornocker and Hash 1981). Averaged across seasons, wolverines were located in alpine fir, alpine fir-spruce, and alpine fir-lodgepole pine cover types 56% of the time, and also in Douglas fir-lodgepole pine (22%) and Douglas fir-larch (17%) (Hornocker and Hash 1981). Copeland et al. (2007) found that use of vegetation communities varied by season but the variation was relatively minor, rock-barren habitats which generally occurred at the highest elevations were generally used at proportions less than their availability except by adult males during summer, and grass-shrub habitat was avoided (Copeland et al. 2007). Modeling of habitat use by Copeland et al. (2007) suggested that in summer, adult females and subadults were associated with whitebark pine, and adult males were associated with rock-barren habitat; in winter, adult females were associated with douglas fir-lodgepole, subadults were associated with douglas fir, and adult males were associated with lodgepole pine. Non-use of Douglas fir-ponderosa pine was consistently indicated in their models, primarily during summer, but it was not strongly significant (Copeland et al. 2007). They noted that lodgepole pine tended to dominate the lower fringes of the subalpine zone and that adult males tended to travel more widely than females, and as such were more likely found in lower, conifer-dominated habitats simply by chance; montane conifer forest accounted for 70% of adult male locations. Copeland et al. (2007) stated: "We think is reasonable to assume that the wolverine's association with particular vegetation types had less to do with the vegetative species than with some other ecological component provided by or associated with a particular habitat. We speculate, as have others, that seasonal variation in habitat use is a response to varying food availability."

Rivers, lakes, mountain ranges, or other topographical features do not seem to block movements of wolverines (Banci 1994, Banci 1988). Considering the wolverine's avoidance of human developments, extensive human settlement and major access routes may function as barriers to dispersal (Banci 1994)

Predation may influence wolverine habitat use, depending on the predator complement in the environment, including humans. In south-central Alaska, wolverine use of rock outcrops was greater than the availability of those areas during summer (Banci 1994), perhaps because rock outcrops were being used as escape cover from aircraft. However, wolverines may have also been hunting marmots and collared pikas (*Ochotona collaris*) (Ibid). In Squaw Valley California a wolverine was observed in a marmot (*Marmota flaviventris*) colony in July 1953 (Ruth 1954). Scat suspected from that animal also contained indigenous squirrel remains (Ibid).

Wolverines may climb trees to escape wolves (Banci 1994), although if the trees are not high enough, such attempts may be unsuccessful (Burkholder 1962). Wolverines are found in a variety of habitats and do not appear to shun open areas where wolves are present. Wolverines occur locally with cougars, especially in British Columbia and the northwestern United States. Trees would not be an effective defense because cougars are adept at climbing. Aside from anecdotal reports, only one report on the use of resting sites by wolverines in forested habitats is published (Banci 1994). Overhead cover may be important for resting sites as well as natal and maternal dens. Resting sites in Montana were often in snow in timber types that afforded cover (Banci 1994).

All studies have shown the paramount importance of large mammal carrion, and the availability of large mammals underlies the distribution, survival, and reproductive success of wolverines. Over most of their range, ungulates provide this carrion, although in coastal areas, marine mammals may be used. Wolverines are too large to survive on only small prey. (Banci 1994). Bone and hide may be important foods. They may be available for several months after an ungulate dies (Banci 1994). Small mammals are primary prey only when carrion of larger mammals is unavailable (Banci 1988).

Porcupines (*Erethizon dorsatum*) occur in wolverine diets in Alaska, the Yukon, and Montana. Although they represent a large meal, porcupines appear to be limited to those wolverines that have learned to kill them (Banci 1988). The frequency of red squirrels (Tamiasciuris hudsonicus) in wolverine diets in northern forested habitats (Gardner 1985; Banci 1988) is a reflection of their wide distribution and availability throughout winter. Arctic ground squirrels (*Spermophilus parryi*) composed 26% of all sciurids in the winter diet of Yukon wolverines (Banci 1988) and the majority of the diet in northwest Alaska, where snowshoe hares were absent (Magoun 1985). Wolverines cache hibernating sciurids such as ground squirrels and hoary marmots (*Marmota caligata*) in the snow-free months for later use and excavate them from winter burrows (Gardner 1985; Magoun 1985). Birds occur in the diet according to their availability. Wolverines prey on ptarmigan (*Lagopus* spp.) in winter in the Yukon (Banci 1988), Alaska, and the Northwest Territories (Banci 1994), Prey that occurs sporadically in diets, such as American marten, weasel (*Mustela* spp.), mink (*M. vison*), lynx, and beaver (*Castor canadensis*), likely are mostly scavenged. Vegetation is consumed incidentally although ungulate rumens and may contain nutrients that wolverines cannot obtain from other foods (Banci 1988).

Although wolverines are mostly scavengers, they can prey on ungulates under some conditions. Because of their low foot loads (pressure applied to substrate) of 22 g/cm2 (Banci 1994), wolverines can prey on larger mammals in deep snow and when ungulates are vulnerable. Caching of food by wolverines has been described by most studies except that in Montana. The frequency of caching by wolverines may be affected in various ways by the presence of other carnivores (Ibid).

The shrinking range of wolverines coincided with that of wolves in the late 1800's and the early 1900's. In some areas, predator control was coupled with the decimation of large mammal populations, such as the northern caribou herds (Banci 1994), reducing food available to wolverines. Wolverines were vulnerable to historical trapping; however, they are also described as being extremely sensitive to the presence of people (Banci 1994).

Starvation likely is an important mortality factor for young and very old wolverines. Suspected deaths from starvation include two young-of-the-year females in southwest Yukon (Banci 1988) and a young female and an old male in Montana (Banci 1994). These animals relied heavily on baits just before their deaths, suggesting that very young and old age classes may be unsuccessful foragers, even if food is abundant (Banci 1994; Banci 1988). Documenting the fates of young males is difficult be-cause of their extensive movements and it is not possible to predict whether sexes differ in their susceptibility to starvation (Banci 1994).

Climate changes are predicted to reduce wolverine habitat and range by 23 percent over the next 30 years, and 63 percent over the next 75 years, rendering remaining habitat significantly smaller and more fragmented. This increased fragmentation and isolation of subpopulations is expected to limit the regular dispersal of wolverines that is necessary to maintain genetic exchange and metapopulation dynamics. Other secondary threats to the wolverine that could work in concert with climate change include harvest, disturbance, infrastructure, transportation corridors, and small effective population sizes. The primary threat of habitat and range loss due to climate change would affect wolverine habitat and, therefore, the magnitude of threats to the wolverine is high (USFWS 2011). If these scenarios are valid, then conservation efforts should focus on maintaining wolverine populations in the largest remaining areas of contiguous habitat and, to the extent possible, facilitating connectivity among habitat patches (McKelvey et al 2011). However climate change has not yet had a detectable effect on the wolverine to this point in time; the threat is nonimminent (USFWS 2011).

In 1977-78, a study conducted by the California Department of Fish and Game did not detect any wolverines in the north Sierras (Hummel 1978). During the winter of 1991-92, the California Deptartment of Fish and Game, University of California Berkeley, and five National Forests conducted a cooperative wolverine study using baited infra-red camera systems at 57 camera stations. Forests involved were the Inyo, Lake Tahoe Basin Management Unit, Shasta-Trinity, Stanislaus, and Tahoe National Forest. No wolverines were detected. During this study, fifteen of the camera units were placed within the Tahoe National Forest for various lengths of time. In 1993 and 1994, the Tahoe National Forest conducted additional studies using five camera units that were monitored by volunteers. No wolverines were detected. During the winters of 1998 through 2004, 136 baited camera survey stations were conducted primarily on the Sierraville Ranger District to USFS Region 5 protocol (Zielinski and Kucera 1995). Wolverines were not detected during these survey efforts.

Anecdotal sightings of wolverines have been reported in the Tahoe National Forest. Ruth (1954) reported a wolverine in Squaw Valley during an Audobon Camp field trip which was observed by about 25 people. Schempf and White (1977) reported three recorded sightings in the Webber Lake area on Sierraville Ranger District, and on Truckee Ranger District near Martis Creek, though these may have been marmots. On Sierraville Ranger District, a wolverine sighting of unknown reliability is recorded near Jackson Meadows Reservoir in 1971, other incidental reports of unknown reliability have come from the public near Webber Lake Falls. More recent incidental sightings that could potentially be wolverine include a 1991 sighting reported in Euer Valley on Truckee Ranger District. A 1992 sighting in the Harding Point area, northeast of Sierraville, was confirmed by track identification. Sightings on the Yuba River Ranger District include one in 1989 in the Haskell Peak area, one in 1990 in the Upper Sardine Lake area, one in 1993 along the Gold Lake Road and Salmon Lake Road area, one in 1998 near Bassett's Station, and one at Sawmill Lake (near Bowman Lake) in 2008. Close to the Tahoe National Forest boundary with Plumas National Forest, sightings have been reported in the Gold Lakes Basin; one around 2006 and one in July 2009. On the Foresthill Ranger District, a wolverine was sighted in the Robinson Flat area in 1980 by a wildlife biologist, in 1992 a wildlife biologist observed a wolverine in the Granite Chief Wilderness Area, and around 2003 a wildlife biologist observed a wolverine near Foresthill. In 2008, following the confirmed detection of the male wolverine in the Tahoe National Forest (discussed in Moriarty et al. 2009), numerous sightings were called in to Tahoe National Forest personnel which vary in date as far back as 1965. These reports include one from Prosser Dam on Truckee Ranger District in August 1965, a report of a dead wolverine found along the Gold Lake Highway in 1972, one from Sawmill Lake in August 1997, one from Bowman Lake Road near Henness Pass in 2000, one in the Webber Lake Falls area in summer 2004, one at Sierra Ski Park in January 2008, and one with unknown year from The Cedars area.

The lone male wolverine that has been documented in the Tahoe National Forest is not known to occur within the project area. All wolverine detections have been concentrated along the Pacific Crest Trail from Hwy 49 south to Hwy 80. Wolverines move to high elevations during the summer, when trail use would be the most prevalent along these trails. Because wolverines are wide ranging, trail re-routes along these small segments of trail would not effect this individual.

B. Wolverine: Effects of the Proposed Action and Alternatives including Project Design Standards

Because this species is so wide ranging, and wolverine populations are not suspected to occur in the Tahoe National Forest, there are no direct, indirect, or cumulative effects that would occur from this project.

C. Wolverine: Conclusion and Determination

It is my determination that implementation of this project will not affect the California wolverine.

PALLID BAT

Status: USFS R5 Sensitive

A. Pallid Bat: Existing Environment

The pallid bat occurs in western North America, from southern British Columbia to central Mexico and east to central Texas (NatureServe 2011). Within its range, it is associated with a variety of low elevation arid communities and at higher elevation conifer communities; its abundance is greatest in dry conditions (Rambaldini 2005). Throughout California, the pallid bat is usually found in low to middle elevation habitats below 6000 feet (Barbour and Davis 1969; Philpott 1997), however, the species has been found up to 10,000 feet in the Sierra Nevada Mountains. The range in California is statewide and it is predicted to occur on every National Forest in the Region (CWHR 2008). Occurrence records from the state (CNDDB 2011) indicate presence within the boundaries of the following National Forests: Cleveland NF, Eldorado NF, Inyo NF, Klamath NF, Lassen NF, Los Padres NF, Modoc NF, Plumas NF, San Bernardino NF, Sequoia NF, Sierra NF, Stanislaus NF, and the Tahoe NF. Forest Service records (NRIS 2011) indicate this species has been observed within the following National Forest boundaries: Angeles NF, Inyo NF, Los Padres NF, Mendocino NF, Modoc NF, Plumas NF, Shasta-Trinity NF, and the Tahoe NF.

Populations have declined in California within desert areas, in areas of urban expansion, and where oak woodlands have been lost (Brown 1996). There is a decreasing trend in abundance in southern California (Miner and Stokes 2005); trends elsewhere in the state have not been assessed. Miner and Stokes (2005) indicate urban expansion into suitable habitat as a source of continued population decline, especially in lower elevation habitat.

Pallid bats mate between October and February, have one to two pups per year, usually from late April to July, which are weaned in August, with seasonal variation (Rambaldini 2005; Hermanson and O'Shea 1983). Maternity colonies disperse after weaning (Rambaldini 2005).

Pallid bats are a gregarious species, often roosting in colonies of 20 to several hundred individuals. Pregnant females gather in summer maternity colonies of up to several hundred females, but generally fewer than 100 (Brown 1996). Males are typically absent from maternal colonies, or living in clusters of males separated from females in caves, mines, or buildings (Barbour and Davis 1969). Mating occurs in October after summer colonies have disbanded (Barbour and Davis 1969). Breeding probably occurs sporadically throughout the winter, at least until the later part of February. As with several other species of bats, live sperm can be retained in the uterus of the female through the winter and fertilize ova as they are released. Gestation period is estimated at 53-71 days (Barbour and Davis 1969). Parturition occurs between May and July with typically two young born (Barbour and Davis 1969; Burt and Grossenheider 1980; Zeiner et al. 1990). Young are weaned in mid to late August with maternity bands disbanding between August and October (Barbour and Davis 1969; Burt and Grossenheider 1980).

Pallid bats are opportunistic generalists that glean a variety of arthropod prey from surfaces, but also capture insects on the wing (Rambaldini 2005). They forage primarily in uncluttered, open habitats (Rambaldini 2005; Ferguson and Azerrad 2004). Pallid bats prey on flightless and mostly ground roving invertebrates, as well as those that perch exposed on vegetation (O'Shea and Vaughn 1977; Hermanson and O'Shea 1983). Common prey species are Jerusalem crickets, longhorn beetles, and scorpions, but it will also forage at low heights of 0.5 - 2.5 meters (1.6-8.2 feet) above the ground on large moths and grasshoppers (Barbour and Davis 1969; O'Shea and Vaughn 1977; Burt and Grossenheider 1980; Philpott 1997; Zeiner et al. 1990).

The pallid bat is strongly associated with arid regions (Hermanson and O'Shea 1983). Low elevation habitat includes rocky, arid deserts and canyons, shrub-steppe grasslands, and karst formations (Rambaldini 2005). It is also found in high elevation conifer forests (ibid.). Miner and Stokes (2005) suggest that riparian, chaparral, oak savannah, and cultivated areas are preferred habitat types, and Baker et al. (2008) further suggest open pine forest within higher elevations. In forested habitats in the Sierra Nevada Mountains, Baker et al. (2008) found pallid bats in areas with greater availability of Sierran mixed conifer and white fir than open meadows, grasslands, barren areas, and montane chaparral. They caution, however that they were unable to discern actual habitat use at a finer scale. Johnston and Gworek (2006) found pallid bat activity in the Sierra Nevada Mountains greatest where there was open mixed conifer forest near short grassland habitat. Roosts located were primarily in incense cedar trees (ibid.).

Day roosts may vary but are commonly found in rock crevices, tree hollows, mines, caves and a variety of man-made structures (Ellison et al. 2003). Tree roosting has been documented in large conifer snags, inside basal hollows of redwoods and giant sequoias, and bole cavities in oaks (ibid.). Cavities in broken branches of black oak are very important and there is a strong association with black oak for roosting. Roosting sites are usually selected near the entrance to the roost in twilight rather than in total darkness. The site must protect bats from high temperatures, as this species is intolerant of roosts in excess of 104 degrees Fahrenheit (Philpott 1997).

Night roosts are usually more open sites and may include open buildings, porches, mines, caves, and under bridges (Barbour and Davis 1969; Philpott 1997; Pierson 1996).

Winter roosts are cool (25-50˚ Fahrenheit) with a stable temperature range, and are located in protected structures, including caves, mines, and buildings (Rambaldini 2005). Pallid bats do not travel far from their summer range to their winter roost location (ibid).

The largest emerging threat to all cave-roosting species is white-nose syndrome. There is a grave concern that it could spread to the western states and California. As of April 2016, White-nose Syndrome.org records detections as far west as Oklahoma, with one reported case in Washington state (https://www.whitenosesyndrome.org/resources/map). This disease has rapidly spread throughout the eastern US and Canada since its discovery in 2006.

Habitat threats include loss of foraging habitat due to urban expansion in low elevation habitat (Philpott 1997; Ferguson and Azerrad 2004; Rambaldini 2005; Miner and Stokes 2005; Pallid Bat Recovery Team 2008) and loss of riparian habitat in arid areas. Conversions of dry grasslands and sagebrush habitat to orchards and other dense vegetative cover reduces foraging habitat (Chapman et al. 1994; Ferguson and Azerrad 2004; Pallid Bat Recovery Team 2008). Pesticide use in these agricultural areas may adversely impact invertebrate populations, thus affecting the pallid bat prey base (Ferguson and Azerrad 2004; Miner and Stokes 2005; Pallid Bat Recovery Team 2008). Intense grazing may likewise adversely impact foraging areas and prey diversity (Ferguson and Azerrad 2004; Ferguson and Azerrad 2004; Pallid Bat Recovery Team 2008), however properly managed grazing may not adversely impact foraging habitat.

The loss of large diameter snags and live trees for roosts due to fire can affect primarily day and night roosts (Miner and Stokes 2005). While this species typically roosts in rock outcrops, it often uses alternate day roosts, which large trees may provide. Retention of existing large trees and long term production of replacement large trees would provide potential habitat into the future.

Mine closures may eliminate roosting sites and hibernacula for pallid bats, even though this species primarily roosts in rock outcrops (Rambaldini 2005; Ferguson and Azerrad 2004; Miner and Stokes 2005; Pallid Bat Recovery Team 2008). Likewise bridge reconstruction may eliminate roost sites if done in a way that does not provide a design suitable to pallid bats (Ferguson and Azerrad 2004).

Pallid bats are also susceptible to disturbance in roosting sites and subsequent displacement (Rambaldini 2005), particularly hibernating individuals. Other human density-related threats include feral cats (Hermanson and O'Shea 1983; Ferguson and Azerrad 2004).

In 1999, Dr. Joe Szewczak, bat researcher from the White Mountain Research Station, initiated a program in the Carman Valley Watershed Restoration area at Knutson Meadow to monitor changes in bat diversity in relation to restoration activities. Pallid bats were detected in Carman Valley on the Sierraville Ranger District of the Tahoe National Forest through these monitoring efforts.

The area considered to analyzing effects to this species include the trail proposals buffered by 0.25 miles. This area would adequately address potential effects of directly disturbing or displacing individuals during trail construction or decommissioning or longer term effects of noise from use of the trails. There are no confirmed pallid bat sightings within the District, but suitable habitat is present. Some foraging habitat may be present in open areas of the project. No roosting habitat such as mine openings, caves, or rock outcrops are present in proximity to the trail.

B. Pallid Bat: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct and Indirect

Because no hibernating, roosting, or natal habitat for this species is present in proximity to the trail, this project would not remove or alter any roosting habitat and no direct effects are expected to occur. This bat forages within open environments for grasshoppers, Jerusalem crickets, and other arthropods. Trail construction would only create an opening four feet wide in a linear pattern, and it would not create open habitats that would change habitat for this species that is biologically meaningful. Therefore indirect effects are not expected to occur.

Cumulative

Because there are no direct or indirect effects to this species, this Project would not add any cumulative effects.

C. Pallid Bat: Conclusion and Determination

It is my determination that implementation of this project will not affect the pallid bat.

TOWNSEND'S BIG-EARED BAT

Status: USFS R5 Sensitive

A. Townsend's Big-Eared Bat: Existing Environment

The Townsend's big-eared bat occurs throughout western North America, from southern British Columbia to central Mexico and east into the Great Plains, with isolated populations occurring in the south and southeastern United States (Pierson and Rainey 1998; Sherwin 1998; NatureServe

2011). In California, the range is nearly state-wide except the highest peaks of the Sierra Nevada Mountains, including each National Forest within Region 5 (CWHR 2008). Records in the California state database also indicate a statewide distribution, and occurrence on most R5 Forests except the Angeles NF, Eldorado NF, Lake Tahoe Basin Management Unit, Los Padres NF, and Six Rivers NF (CNDDB 2011). Forest Service NRIS records (accessed November 2011) since the 1990s are more scant, and indicate a presence on the Cleveland NF, Eldorado NF, Inyo NF, Mendocino NF, Modoc NF, Plumas NF, Shasta-Trinity NF and Lake Tahoe Basin Management Unit. Townsend's big-eared bats have been captured during survey efforts on the Six Rivers NF (Siedman and Zabel 2001).

Historically, the Townsend's big-eared bat was found throughout California as a scarce, but widespread species (Barbour and Davis 1969). It ranges from sea level to 3,300 meters (10,827 feet) in elevation in a wide range of vegetation types (Sherwin 1998; Barbour and Davis 1969; Philpott 1997; CWHR 2008). Its distribution is strongly correlated to geomorphic features such as natural and man-made caves, buildings, and bridges (Pierson et al. 1999; Ellison et al. 2003a and 2003b; Sherwin et al. 2003; Gruver and Keinath 2006). Caves and mine adits typically are used as hibernacula by both sexes (Piaggio 2005). These, along with old buildings, bridges, and large trees may be used as roost sites (Piaggio 2005). It is generally understood that these bats have high roost sight fidelity (Pierson and Rainey 1998; O'Shea and Bogan 2003; Ellison et al. 2003a and 2003b; Piaggio 2005; Gruver and Kenaith 2006), even though use of a specific sight may vary through time (Sherwin et al. 2003).

Population trends have been reportedly declining across the state (Pierson and Rainey 1998; Pierson et al. 1999; Miner and Stokes 2005). However, recent research suggests that absence at historic sites, and subsequent reports of population declines, may be a result of insufficient survey effort, especially where multiple potential roosts are available (Ellison et al. 2003a; Sherwin et al. 2003). Furthermore, statistical inferences about this species cannot be made for populations in the western United States (Ellison et al. 2003a). As such, Ellison and others (2003a) suggest caution in interpreting the population declines in California. It is clear that further research and population monitoring is needed to determine trends in the state.

With the above caution in mind, Pierson and Rainey (1998) reported on Townsend's big-eared bat populations in California. They reported substantial changes over the last 40 years in Townsend's big-eared bat total individuals (54 percent decline), maternity colonies (52 percent decline), available roosts (45 percent decline), and average colony size (33 percent decline). Pierson and others (Pierson and Rainey 1998; Pierson et al. 1999) did report that there was unmistakable evidence that some roosts were deliberately destroyed and bats were killed. The Mother Lode country (central Sierra Nevada Mountains and foothills) and the Colorado River area apparently have the most marked declines (Pierson and Rainey 1998).

Mating typically occurs from November to February after bats have entered their hibernaculum for the winter, although some females will be inseminated prior to hibernation (Barbour and Davis 1969; Burt and Grossenheider 1980; Jameson and Peeters 1988; Kunz and Martin 1982; Zeiner et al. 1990). After delayed implantation and a 56-100 day gestation period females give birth to a single pup in May or June (ibid.), sometimes after a move to a nursery colony (Pierson and Rainey 1998). In western North America, almost all spring and summer concentrations of Townsend's big-eared bats are females that have returned to their natal site to give birth and raise their young (Pierson and Rainey 1998). Young are weaned in six weeks, and can fly two-and-a-half to three weeks after birth (Barbour and Davis 1969; Burt and Grossenheider 1980; Jameson and Peeters 1988; Kunz and Martin 1982; Zeiner et al. 1990). Caves and mine adits are commonly used as maternity sites, as well as for winter hibernacula. Males leave the nursery colony after the first summer and typically roost alone (Pierson and Rainey 1998).

Townsend's big-eared bats hibernate singly or in small clusters, usually several dozen or fewer, from October to April (Zeiner et al. 1990). Winter hibernating colonies are composed of mixed-sexed groups and may range from a single individual to several hundred animals (Piaggio 2005). *P. townsendii* hibernates throughout its range in caves and mines where temperatures are approximately 50 degrees Farnheit or less, and generally above freezing. Individuals may move during winter in response to temperature change (Barbour and Davis 1969). Townsend's big-eared bats utilize well-ventilated, cold caves and mine tunnels as hibernacula, in particular locations from which they can hang from the ceiling (Gruver and Keinath, 2006; Pierson and Rainey 1998). In addition to caves and mine tunnels, bridges and old buildings may be utilized as roosts (Barbour and Davis 1969; Pierson and Rainey 1998). Roosting in tree hollows has been reported in coastal California habitats (Fellers and Pierson 2002; Gellman and Zielinski 1996).

Townsend's big-eared bats do not migrate long distances (Barbour and Davis 1969; Humphrey and Kunz 1976; Dobkin et al. 1995; Woodruff and Ferguson 2005). Townsend's big-eared bats change roosts throughout the season (Fellers and Pierson 2002; Sherwin et al. 2001), which may complicate survey efforts (Sherwin et al. 2003). Even in cool climates, Townsend's big-eared bats may change roosts in the winter (Woodruff and Ferguson 2005).

This species is a moth specialist but feeds on a variety of lepidopterans (i.e., moths, butterflies, skipper butterflies, and moth-butterflies) (Pierson and Rainey 1998). Pierson et al. (1999) summarized other research that includes consumption of other invertebrate orders in small amounts. Small moths, beetles, and a variety of soft-bodied insects also are taken in flight using echolocation, or by gleaning from foliage (Jameson and Peeters 1988; Zeiner et al. 1990). They are known to drink water. This bat forages relatively close to its roosts sites (Gruver and Kenaith 2006).

Flight is slow and maneuverable, with the species capable of hovering (Zeiner et al. 1990; Gruver and Kenaith 2006) and perhaps gleaning insects off foliage (Gruver and Kenaith 2006). Foraging usually begins well after dark (Kunz and Marten 1982). This bat will forage above and within the canopy (Pierson et al. 1999), often along forest edges and riparian areas (Piaggio 2005), and seems to be well adapted to a moderately cluttered canopy (Gruver and Kenaith 2006). As stated, foraging habitat includes a wide variety of vegetation types. Suitable foraging habitat in California includes agricultural types, dense forests, desert scrub, moist coastal forests, oak woodlands, and mixed conifer-deciduous forests (Pierson and Rainey 1998), in particular along habitat edges (Fellers and Pierson 2002). Habitat connectivity between roosting and foraging sites may be important for this species, especially because individuals tend to avoid open spaces (Gruver and Kenaith 2006).

This bat is associated with a wide range of vegetative types, including forests, desert scrub, pinyon-juniper woodlands, and agricultural development (Gruver and Keinath 2006; Kunz and Martin 1982; Piaggio 2005; CWHR 2008). Roost structure is believed to be more important than the local vegetation (Gruver and Keinath, 2006; Pierson and Rainey 1998) and the presence of suitable caves or cave-like structures defines the distribution of this species more so than does suitable foraging habitat (Barbour and Davis 1969; Pierson and Rainey 1998; Piaggio 2005; Gruver and Keinath, 2006). In California, this bat is known to use lava tubes, man-made structures (buildings, bridges, and mines, for example), some limestone caves (Kunz and Martin 1982), and large trees (Piaggio 2005).

Wildlife habitat associations in California (CWHR 2008) are broad. These bats are often associated with forest edges, open forests, shrub and scrub habitats, grasslands, and riparian areas (CWHR 2008; or drier habitats where there is free water (Geluso 1978). Free water is an important habitat feature for this bat as it has a relatively poor urine-concentrating ability; it can meet some of its water needs metabolically (Geluso 1978).

The most critical habitat feature for this species is cave and cave-like roosting structures and hibernacula. With the increase in mining in the 1800s, potential roosting sites increased with the development of mines. Actual use of these mines for roost purposes likely did not occur until many of the mines shut down or were abandoned. This species is found in mines more than any other species (Barbour and Davis 1969; Sherwin et al. 2003). Likewise, buildings and bridges are also used by Townsend's big-eared bats, especially on the west coast and in forested areas (Barbour and Davis 1969). Human disturbance in caves and mines can result in the bats moving their roosting location within the cavern or abandoning the site altogether (Barbour and Davis 1969; Pierson et al. 1999; Gruver and Keinath 2006).

The largest emerging threat to all cave-roosting species is white-nose syndrome. There is a grave concern that it could spread to the western states and California. As of April 2016, White-nose Syndrome.org records detections as far west as Oklahoma, with one reported case in Washington state (https://www.whitenosesyndrome.org/resources/map). This disease has rapidly spread throughout the eastern US and Canada since its discovery in 2006.

A significant threat to Townsend's big-eared bats is disturbance or destruction of roost sites, in particular hibernacula and nursery sites (Pierson et al. 1999; Piaggio 2005; Woodruff and Ferguson 2005; Bradley et al. 2006). Roost structure is believed to be more important than the local vegetation (Gruver and Keinath, 2006; Pierson and Rainey 1998) and the presence of suitable caves or cave-like structures defines the distribution of this species more so than does suitable foraging habitat (Barbour and Davis 1969; Pierson and Rainey 1998; Piaggio 2005; Gruver and Keinath, 2006). Visitation during critical periods can adversely affect bats in those sites, often leading to reduced populations (Pierson et al. 1999). In such an event, rousing from torpor uses valuable fat reserves which are needed to sustain physiological processes throughout the hibernation period. A single visit may result in abandonment of the roost (Barbour and Davis 1969; Zeiner et al. 1990). Low fecundity (one pup/year) and high first year mortality means disturbance at a hibernacula or nursery roost can be potentially detrimental, although by limiting disturbance, populations can recover in part because survival rate in subsequent years is higher (Pierson et al. 1999).

Mine closures, often with the intent to protect human safety, can eliminate access to roosts and hibernacula (Miner and Stokes 2005). Reactivation of mines may eliminate cave roosts and hibernacula, or cause disturbance such that bats will abandon a site (Pierson et al. 1999). Because this species uses alternate roost sites over time (during a single season as well as over many years), potential roosts must be surveyed at least eight times in order to determine vacancy (Sherwin et al. 2005).

Reopening of closed or inactive mine tunnels for mineral extraction would likely disturb all roosting bats, resulting in abandonment of those sites. If many tunnels in relatively close proximity are reopened, there could be serious adverse impacts to bat roost sites, and subsequently to bat populations at the local scale because this species, among others, has a high affinity to roost sites. Present day mining operations are likely to be surface or open-pit efforts that would affect foraging habitat (vegetation) as well as the tunnels that have become roosting habitat (Bogan 2000).

Contaminants may come from various sources and may directly or indirectly affect bats, although little research has been done (O'Shea et al. 2000). Waste material impoundments can be a threat to this species because it must drink water, and water sources are especially important in dry habitats; bats have been killed when trapped in oily water associated with drilling operations (O'Shea et al. 2000). Radiation may be a source of toxicity for bats roosting deep in mines, as well (ibid.). Pesticide spraying may locally deplete food resources which may be particularly challenging for nursing females that need to forage further and further from nursery roosts as pups grow (Woodruff and Ferguson 2005).

The effects of timber management and prescribed fire on bat habitat are not well understood (Woodruff and Ferguson 2005). Humes et al. (1999) found bats to be more active in old-growth and thinned forest stands than in dense, unthinned stands, suggesting that the increased structural diversity benefitted bats, including Townsend's big-eared bats.

In the Tahoe National Forest, the only documented maternal colony of Townsend's big-eared bats occurs near the town of Sierra City, approximately 28 miles to the north of the treatment units. Townsend's big-eared bats were also observed on the Tahoe National Forest in Carmen Valley by Dr. Joe Szewczak (White Mountain Research Station) between 1999 and 2001.

There are no mine openings, abandoned buildings, or caves within the project area that would provide roosting or hibernating habitat for this species. This species could utilize snags located within or adjacent to the trail.

B. Townsend's Big-Eared Bat: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct and Indirect

If this species is using any snags along the trail during trail construction, they could be disturbed during project implementation. The project design standards include a requirement to locate the trail where practicable to avoid the need to fall large trees and snags, or those displaying wildlife use (cavities, nests). This would reduce the probability of directly affecting bats.

This bat gleans insects (especially moths) from vegetation, especially riparian vegetation and hardwoods. Constructing the trail would remove a small amount of vegetation. It would not alter the overstory canopy cover, or remove foraging habitat enough that would biologically meaningful to this species. Therefore, indirect effects to habitat are not expected to occur.

Cumulative

Cumulative effects to this species occurs primarily from human disruption of colonies at roosts located in caves and mines and old buildings. This project may add some negative effects to this species if bats are roosting in a snag adjacent to recently dead trees that are removed during trail construction or maintenance. No colonies are known to occur within the vicinity of where the trail would be constructed. Because this project is not likely to disturb, disrupt, or negatively affect entire colonies of this species, any additional cumulative effects from this project are considered to be minimal.

C. Townsend's Big-Eared Bat: Conclusion and Determination

It is my determination that implementation of this project may affect individuals, but is not likely to result in a trend toward Federal listing or loss of viability for the Townsend's big-eared bat within the planning area of the Tahoe National Forest. In the absence of a range wide viability assessment, this viability determination is based on local knowledge of the Townsend's big-eared bat as discussed previously in this evaluation, and professional judgment".

FRINGED MYOTIS

A. Fringed Myotis: Existing Environment

The fringed myotis is found in western North America from south-central British Columbia to central Mexico and to the western Great Plains (Natureserve 2012). In California, it is distributed statewide except the Central Valley and the Colorado and Mojave Deserts (CWHR 2008).

In California, the species is found the throughout the state, from the coast (including Santa Cruz Island) to greater than 5,900 feet in elevation in the Sierra Nevada. Records exist for the high desert and east of the Sierra Nevada (e.g., lactating females were captured in 1997 by P. Brown near Coleville on the eastern slope of the Sierra Nevada). However, the majority of known localities are on the west side of the Sierra Nevada (Angerer and Pierson draft). Museum records suggest that while the fringed myotis is widely distributed in California, it is rare everywhere.

According to Forest Service records, the fringed myotis is found on the Angeles NF, Eldorado, NF, Los Padres NF, Mendocino, NF, Modoc NF, Plumas, NF, Shasta-Trinity, NF, the Sierra NF, Lake Tahoe Basin Management Unit, and the Tahoe NF. State records (CWHR 2008) add the Cleveland NF, Inyo NF, Klamath NF, Lake Tahoe Basin, Lassen NF, San Bernardino NF, Sequoia NF, Six Rivers NF, and Humboldt-Toiyabe NF.

The fringed myotis roosts in crevices found in rocks, cliffs, buildings, underground mines, bridges, and in large, decadent trees (Weller 2005). The majority of maternal roost sites documented in California have been found in buildings (Angerer and Pierson draft). Mines are also used as roost sites (Cahalane 1939, Cockrum and Musgrove 1964, Barbour and Davis 1969). Like many cave roosting species, fringed myotis colonies are susceptible to disturbance in hibernacula and maternal colonies (CWHR 2008; O'Farrell and Studier 1973, 1980).

Maternity roosts have been found in sites that are generally cooler and wetter than is typical for most other Vespertilionids (Angerer and Pierson draft). Recent radio-tracking studies in the forested regions of northern California have shown that this species forms nursery colonies in predominantly early to mid- decay stage, large diameter snags from 23” to 66” dbh (Weller and Zabel 2001).

The fringed myotis occurs in dry woodlands (oak and pinyon-juniper most common (Cockrum and Ordway 1959, Jones 1965, O’Farrell and Studier 1980, Roest 1951), hot desert-scrub, grassland, sage-grassland steppe, spruce-fir, mesic old growth forest, coniferous and mixed deciduous/coniferous forests, including multi-aged sub-alpine, Douglas fir, redwood, and giant sequoia (O’Farrell and Studier 1980, Pierson and Heady 1996, Pierson et al. 2006, Weller and Zabel 2001). To generalize, this species is found in open habitats that have nearby dry forests and an open water source (Keinath 2004).

There seems to be increased likelihood of occurrence of this species as snags greater than 11.8 inches in diameter increases and percent canopy cover decreases (Keinath 2004). Large snags and low canopy cover, typical of mature forest habitat types, offer warm roost sites (Keinath 2004). Decay classes were two to four (Keinath 2004) in ponderosa pine, Douglas-fir, and sugar pine. Water sources may include artificial sources, such as stock tanks and ponds, in addition to natural sources (Keinath 2004).

Home range size varies with insect abundance, increasing as the number of available insects decreases. Keinath (2004) reports study averages of about 100 acres. Little is known about predation, but it is not suspected to significantly affect fringed myotis populations (Keinath 2004).

Fringed myotis appears to be highly dependent on tree roosts within forest and woodland habitats and potentially requires denser vegetation for foraging. In some forested settings, fringed myotis appears to rely heavily on tree cavities and crevices as roost sites (Weller 2005), and may be threatened by certain timber harvest practices. For example, Chung-MacCoubrey (1996) in Arizona found that this species prefers large diameter (18-26 inch dbh) conifer snags. Most of the tree roosts were located within the tallest or second tallest snags in the stand, were surrounded by reduced canopy closure, and were under bark (ibid.). Tree roosting behavior is consistent with an observed association between fringed myotis and heavily forested environments in the northern part of its range (M. Brigham pers. comm., E. Pierson and W. Rainey pers. obs.).

This species shows high roost site fidelity (O’Farrell and Studier 1980), especially when roost structures are durable or in low availability (Brigham 1991, Kunz 1982, Lewis 1995). Weller and Zabel (2001) noted frequent roost switching in tree roosts, but high fidelity to a given area. Roost switching has also been reported for caves (Baker 1962) and buildings (O’Farrell and Studier 1973, Studier and O’Farrell 1972). Fringed myotis are highly sensitive to roost site disturbance (O’Farrell and Studier 1973, 1980).

This species often forages along secondary streams, in fairly cluttered habitat. It also has been captured over meadows (Pierson et al. 2001). Fringed myotis is known to fly during colder temperatures (Hirshfeld and O'Farrell 1976) and precipitation does not appear to affect emergence (O'Farrell and Studier 1975). Post-lactating females have been known to commute up to 13 kilometer (8 miles) with a 930 meter (3,100 feet) elevation gain between a roost and foraging area (Miner and Brown 1996). Keinath (2004) found that travel distances from roosting to foraging areas may be up to five miles.

The removal of snags and hardwoods during timber harvesting and the loss of hardwoods through conifer and brush competition (from a lack of fire management) has caused reductions for both roosting structures and foraging habitat. These practices are likely to be more severe on privately owned lands. An increased demand for firewood can also lead to a decrease in available snags as roosts. Habitat alteration threatens this species because it is dependent on older forest types. Keinath (2004) summarized this in the Rocky Mountain Region conservation assessment for the fringed myotis, indicating that this species depends on abundant large diameter snags and trees with thick loose bark.

The largest emerging threat to all cave-roosting species is white-nose syndrome. There is a grave concern that it could spread to the western states and California. As of April 2016, White-nose Syndrome.org records detections as far west as Oklahoma, with one reported case in Washington state (https://www.whitenosesyndrome.org/resources/map). This disease has rapidly spread throughout the eastern US and Canada since its discovery in 2006.

Fringed myotis are known to occur on the Tahoe National Forest and have been detected in Carman Valley (Szewczak 2004), Antelope Valley (California Fish and Wildlife CNDDB) and 3 miles southwest of Downieville, California (Forest Service NRIS Database). The nearest of these detections to the analysis area are the occurrences southwest of Downieville. Surveys have not been conducted within the analysis area.

Bat acoustic monitoring and mist-netting conducted in association with several projects have identified this species present in the following locations: mine opening approximately 2 miles SW of Downieville, CA; within meadows and riparian areas associated with the following Reservoirs: Nevada County: Spaulding Reservoir, Nevada Co.; Deer Creek Forebay, Nevada Co.; Milton Reservoir, Sierra Co.

Older snags within the project area could provide roosting habitat for this species.

B. Fringed Myotis Bat: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct and Indirect

There are no rocky outcrops, known mine openings or abandoned buildings within units that could support roosting for this species. This bat species forms nursery colonies in predominantly early to mid- decay stage, large diameter snags from 23" to 66" dbh (Weller and Zabel 2001). Older snags within the project area could provide roosting habitat for this species. If this species is using any snags along the trail during trail construction, they could be disturbed during project implementation. The project design standards include a requirement to locate the trail where practicable to avoid the need to fall large trees and snags, or those displaying wildlife use (cavities, nests). This would reduce the probability of directly affecting bats.

Constructing the trail would remove a small amount of vegetation. It would not alter the overstory canopy cover, or remove foraging habitat enough that would biologically meaningful to this species.

Cumulative

Cumulative effects to this species occur from the use of pesticides that decrease insect prey populations, disturbances at roosts within caves, removal of abandoned buildings that are used for maternity sites, and the loss of riparian hardwoods. This project may add some negative effects to this species if bats are roosting in a snag adjacent to recently dead trees that are removed during trail construction or maintenance. No colonies are known to occur within the vicinity of where the trail would be constructed. Because this project is not likely to disturb, disrupt, or negatively affect entire colonies of this species, any additional cumulative effects from this project are considered to be minimal.

C. Fringed Myotis Bat: Conclusion and Determination

It is my determination that implementation of this project may affect individuals, but is not likely to result in a trend toward Federal listing or loss of viability for the fringed myotis within the planning area of the Tahoe National Forest. In the absence of a range wide viability assessment, this viability determination is based on local knowledge of the species as discussed previously in this evaluation, and professional judgment".

Aquatic Species

CALIFORNIA RED-LEGGED FROG

Status: USFWS Threatened

A. California Red-Legged Frog: Existing Environment/Environmental Baseline

On June 24, 1996, the California red-legged frog, was listed as federally threatened (USDI Fish and Wildlife Service 1996). The Final California Red-legged Frog Recovery Plan was released on September 12, 2002 (USDI Fish and Wildlife Service 2002; 67 FR 57830). On March 17, 2010, the USFWS finalized designation of critical habitat that includes three locations in or adjacent to the Tahoe National Forest (USDI Fish and Wildlife Service 2010; 75 FR 12816). Locations include PLA-1, Michigan Bluff; NEV-1, Sailor Flat; and YUB-1, Oregon Creek (Figure 1). The recovery objective is to reduce threats and improve the population status of the California red-legged frog sufficiently to warrant de-listing. The strategy for recovery includes protecting existing populations by reducing threats, restoring and creating habitat that will be protected and managed in perpetuity, surveying and monitoring populations, conducting research on the biology of the species and threats to the species, and re-establishing populations of the species within the historic range.

The Recovery Plan for the California Red-legged Frog (USDI Fish and Wildlife Service 2002) indicates that current and historic distribution of the species is west of the Sierra-Cascade crest. Two recovery units in the Sierra Nevada foothills overlap with the Tahoe National Forest (TNF) as follows (75FR12816, US Fish and Wildlife Service, 2010):

NEV-1, Sailor Flat: 8,285 acres: 3,171 ac. federal (1,577 TNF); 12 ac. State; 5,102 ac. Private
PLA-1, Michigan Bluff: 814 Federal (814 TNF); 430 ac. Private.

A third Recovery Unit, YUB-1, overlaps with the Plumas National Forest, just west of the Tahoe National Forest, with Bullards Bar Reservoir forming its eastern boundary.

The California red-legged frog is a highly aquatic species typically found in cold water ponds and stream pools with depths exceeding 0.7 meters and with overhanging vegetation such as willows, as well as emergent and submergent vegetation (Hayes & Jennings 1988). It is generally found at elevations below 4,000 feet, but has been found above this (Martin 1992). Barry and Fellers (2013) report the elevational range of most of the historical localities as 200-900m (656 to 2953 feet); and that three apparently extirpated populations at 1500 to 1536 m elevation in Yosemite may have originated from deliberate translocations. Additionally, they report that "historically, *R. draytonii* in the Sierra Nevada probably bred in stream pools, which tend to be small with limited forage and thus may have constrained the historical size and number of Sierra Nevada *R. draytonii* populations." Barry and Fellers (2013) report that twelve historical Sierra Nevada and Cascades locality records were narrow permanent or near-permanent creeks, typically within a few hundred meters of the headwaters, as are three recent *R. draytonii* occurrences, and that this suggests that this species uses stream habitat in the Sierra Nevada in the same manner as do Coast Range populations. They found that the current range of 330 to 1,020 m elevation (1,083 to 3,346 feet) differs little from the historical range (Barry and Fellers 2013).

This frog is generally found in or near water but does disperse away from water after rain storms (Martin 1992). This species of frog breeds along aquatic vegetation in deep, slow water (<2% gradient) environments during the months of November through March in most of their current range (USDI Fish and Wildlife Service 1996). Breeding in the Sierra Nevada would occur later due to freezing temperatures between November and February. Breeding would likely occur between March and May at higher elevations (Freel 1997, personal communication). Permanent or nearly permanent pools are required for tadpole development, and emergent and overhanging vegetation is used as refugia by adult frogs. Ponds with cattails or other emergent vegetation provide good cover (Martin 1992). The amount of time to metamorphosis is highly dependent on temperature (Calef 1973). Tadpole development takes 11 to 20 weeks (Storer 1925, Calef 1973). Water quality is also very important. Adult frogs normally become sexually mature in two (males) to three (females) years and can live as long as ten years or more.

The California red-legged frog requires permanent aquatic habitats for breeding, feeding and shelter. As adults, they may also utilize moist, sheltered, terrestrial habitats near streams. In the proposed ruling to list this species, the United States Fish and Wildlife Service cites Rathbun et al. (1993) in reporting that this frog estivates in small mammal burrows and moist leaf litter up to 85 feet from water in dense riparian vegetation. This behavior occurs where the aquatic habitat is intermittent in nature. During wet periods, especially in the winter and early spring months, California red-legged frogs disperse away from breeding habitat to seek suitable foraging habitat. This dispersal behavior can result in California red-legged frogs ending up in isolated aquatic habitats as far as one mile from their natal pond.

In a California red-legged frog telemetry study at Hughes Pond, in the Sierra Nevada, Tatarian and Tatarian tracked the movement and dispersal patterns of the California red-legged frog between 2004 and 2007 at an ephemeral pond (Hughes Pond, Butte County) in a conifer-hardwood ecosystem in the Sierra Nevada. This work found high site fidelity to the pond and seep area. In years when the pond dried, all radio-tagged frogs moved downstream and downslope to a seep area. In years the pond was perennial, no frogs moved away from the pond. Frogs initiated movements when the pond was dry and after the first 0.5 cm. of rain in the fall. Individuals typically moved aquatically 105 linear meters (345 ft.) from the source pond, while one male moved aquatically 208 meters (682 ft.).

Ideal breeding habitat of California red-legged frogs is characterized by dense, shrubby riparian vegetation associated with deep (≥ 2 feet), still or slow-moving water (Jennings 1988, Hayes and Jennings 1988). The shrubby riparian vegetation that structurally seems to be most suitable for California red-legged frogs is that provided by willow, cattails and bulrushes (Jennings 1988). However, California red-legged frogs have been found in less than ideal habitats and a combination of these factors is more important than an individual habitat component (Hayes and Jennings 1988). Small to medium perennial streams can also provide breeding habitat if the streams are not subjected to scouring flows during egg development. Streams in this category generally have the potential for deep pools and riparian vegetation to provide the habitat requirements for this frog. Permanent or nearly permanent pools that hold water into the summer are required for tadpole development. Emergent and overhanging vegetation is used as a brace for egg deposition and as cover by adult frogs.

While California red-legged frogs are generally found in or near water, during periods of wet weather, starting with the first rains of fall, individual frogs may make overland excursions through upland habitats (USDI Fish and Wildlife Service 2002, pg.12). Various studies of California red-legged frog movement suggest that frog movement often occurs up to 1 mile, and one study showed frogs moving up to 2 miles without apparent regard to topography, vegetation type, or riparian corridors (USDI 2002, pgs 12-13). The Tahoe National Forest considers potential suitable breeding habitat on National Forest lands within 1 mile of a project unit to be a potential source of frogs into the project unit area.

Dispersal habitat generally includes moist, shaded areas with vegetation that provides cover. However, individuals may move through areas that could be considered to be unsuitable for frogs. Normally, frogs travel along riparian corridors and can be found adjacent to streams, meadows or marsh areas. Adults feed primarily on aquatic and terrestrial invertebrates, but large adults will eat small rodents such as deer mice.
This species is highly restricted in the Sierra Nevada, and has been eliminated from 75% of its historic range (Jennings 1992). Habitat loss and alteration, the introduction of bullfrogs and other

aquatic predators, and historic timber harvest have been implicated in the population decline (Jennings 1988, Moyle 1973).

In amending the Forest Plans of Forests in the Sierra Nevada, the Sierra Nevada Forest Plan Amendment (USDA Forest Service 2001, 2003), established Critical Aquatic Refuges. Critical Aquatic Refuges are small subwatersheds that contain either:

- Known locations of threatened, endangered or sensitive species,
- Highly vulnerable populations of native plant or animal species, or
- Localized populations of rare native aquatic- or riparian-dependent plant or animal species.

The Tahoe National Forest has currently has two Critical Aquatic Refuges--Upper Independence Creek and the Sierra Buttes. Both of these were established for other aquatic species, and they occur outside of the range of the California red-legged frog.

<u>Designated Critical Habitat Primary Constituent Elements:</u>

1. Aquatic breeding habitat which consists of standing bodies of fresh water including natural and manmade (e.g. stock) ponds, slow-moving streams or pools within streams, and other ephemeral or permanent water bodies that typically become inundated during winter rains and hold water for a minimum of 20 weeks in all but the driest years.
2. Aquatic non-breeding habitat which consists of freshwater pond and stream habitats as described for breeding habitat except they may not hold water long enough for the species to complete its aquatic life cycle, but these water bodies still provide shelter, foraging, predator avoidance and aquatic dispersal of juvenile and adult frogs. Examples include but are not limited to intermittent creeks, seeps, quiet water refugia within streams during high water flows, and springs of sufficient flow to withstand short term dry periods.
3. Upland areas adjacent to or surrounding breeding and non-breeding aquatic and riparian habitat up to a distance of 1.0 mile. Including various vegetational types such as grassland, woodland, forest, wetland or riparian areas that provide shelter, forage and predator avoidance.
4. Dispersal Habitat which includes upland or riparian areas within and between occupied or previously occupied sites that are located within 1.0 mile of each other. These sites support movement between sites and include various natural habitats.

Potential risk factors to the California red-legged frog from resource management activities include modification or loss of habitat or habitat components, primarily aquatic and adjacent riparian environments used for reproduction, cover, foraging, and aestivation. Egg survival can be impacted by mining and road/trail construction through increases in fine sediments. Livestock grazing directly affects riparian vegetation, emergent vegetation, causes nutrient loading, and also affects channel morphology and hydrology. Timber harvest can result in loss of riparian vegetation and increased erosion and siltation of aquatic habitats (USDA Forest Service 2001). The fungal pathogen *Batrachochytrium dendrobatidis* (*Bd*) is reported in California red-legged frogs and bullfrogs (Padgett-Flohr and Hopkins 2010, P. Tatarian and G. Tatarian 2010, Fellers 2011), and in foothill yellow-legged frogs (Rachowicz et al. 2006, P. Tatarian and G. Tatarian 2010). Crayfish (McMahon et al. 2012) can act as an intermediate host (McMahon et al. 2012), as well as bullfrogs (*Rana catesbeiana*) and treefrogs (*Pseudacris regilla*) (Padgett-Flohr and Hopkins 2009). Tatarian and Tatarian (2010) demonstrated relatively high levels of *Bd* infection in *Rana draytonii* in the Sierra Nevada foothills, from four sampled populations--62% at Big Gun Diggings (Placer 1 Critical Habitat Unit), 100% at Sailor Flat (Nevada 1 Critical Habitat Unit), 50% at Spivey Pond (Eldorado 1 Critical Habitat Unit), and 29 to 71% at Hughes Pond (Butte 1 Critical Habitat Unit) (USDI 2010).

Although *R. draytonii* are susceptible to *Bd* infection, they remain asymptomatic, and there may be a strong selection for *Bd* resistance (Padgett-Flohr, 2008). Padgett-Flohr and Hopkins (2010) found that *Bd* occurrence within a landscape was spatially related, where ponds within approximately 1000-1500 m of *Bd* hotspots were more likely to test positive. They did not find any influence on the *Bd* status of a pond from local land use, such as the presence or absence of grazing or recreational activity (fishing, hiking, and trail riding) and developed lands, indicating that livestock and recreational activity are not transporting *Bd* between ponds (Padgett-Flohr and Hopkins, 2010).

Conservation Recommendations (USDI Fish and Wildlife Service 2001) that may be applicable to Tahoe National Forest management activities include:

1. Assist the USFWS in implementing recovery actions identified within the Draft Recovery Plan for the red-legged frog, including:
    a. Working with the USFWS and other interested parties in developing a reestablishment program for red-legged frogs on National Forest Land.
    b. Developing a nonnative predator (e.g., bullfrogs and warm water fish spp.) eradication program.

2. Any individuals handling red-legged frogs should be prior-approved by the USFWS. All trapping protocol utilized should be prior-approved by the USFWS;

3. Prior to activities within Core Areas identified in the California Red-legged Frog Recovery Plan, a Landscape Analysis should be completed and submitted for approval by the Service. The Landscape Analysis should include, but not be limited to:
    a. Discussions of the management and maintenance in perpetuity of the habitats for red-legged frogs.
    b. Discussions of runoff control and maintenance of hydrology of the aquatic habitat.
    c. Provisions for the design and implementation of a bullfrog eradication program for all aquatic areas.
    d. Provisions for management and maintenance of upland habitat within the Core Areas.
    e. Provisions for a written report to the USFWS, and CDFG on the functioning of the Core Areas five years after the completion of the Landscape Analysis. The report should recommend maintenance practices, repairs, *etc*., (subject to review and approval by the USFWS and CDFG) necessary to ensure the continued functioning of Core Areas as red-legged frog habitat.

4. At least 80 percent of natural streambank stability should be maintained at the end of the authorized grazing season in areas that are occupied by red-legged frogs or red-legged frog habitat within CARs. This means that no more than 20 percent of the natural streambank stability could be altered by activities such as, but not limited to: livestock trampling, chiseling and sloughing, OHV use, stream crossings, and recreational use.

5. Encourage or require the use of appropriate California native species in revegetation and habitat enhancement efforts associated with projects authorized by the Forest Service.

Within the Tahoe National Forest, suitable California red-legged frog breeding habitat is considered to include west slope aquatic habitats within the species' elevational range, especially ponds, lakes, and pools where water persists through July in years with average precipitation. Breeding adults are often associated with deep (greater than 0.7 meter) still or slow moving water (Hayes and Jennings 1988). Higher quality sites contain: water depth greater than 0.7 meters for breeding that lasts through the end of July, still or slow moving water, water temperatures between 9 to 21 degrees Celsius (48 to 70 degrees Fahrenheit) for egg laying and larval development (Nussbaum et al. 1983 *in* US Fish and Wildlife Service 2002), and emergent aquatic vegetation or woody debris for egg deposition braces. Because of these associations, steeper (>4 percent), boulder-driven streams that receive peak runoff flows from snowmelt during May or June, are less likely to support this species, as are smaller exposed ponds and pools where temperatures exceed 22 degrees Celsius. Although the presence of non-native predators (bullfrogs, sunfish, etc.) may reduce habitat quality, it does not necessarily preclude the presence of California red-legged frogs.

Since 1994, Tahoe National Forest biologists have regularly noted amphibians found in aquatic habitats and annually conduct stream surveys for aquatic species across portions of the forests. Suitable habitat such as marshes, ponds and low gradient streams occur on a number of sites within the historical range of this species in the Forest. Within suitable habitat, most of these surveys have followed the USFWS California red-legged frog survey protocol (1997, 2005). From 1996 through 1998 herpetologists from the California Academy of Sciences conducted extensive field surveys throughout the Forest, documenting all reptiles and amphibians (Vindum et al. 1997, Vindum and Koo 1998, Vindum and Koo 1999). In 1997, Dr. Gary Fellers, USGS, and Kathleen Freel surveyed 79 sites between May 20 and September 22, selected with the goal of visiting all suitable sites on the Forest which could reasonably support red-legged frogs. Dr. Fellers concluded "In general, there did not appear to be much habitat which was suitable for this frog on Forest Service lands" (Fellers, 1997). Those surveys followed the survey protocol of Fellers and Freel (1995). To date, no red-legged frogs have been found to occur in the Forest.

In 2007, Fellers and Kleeman (2007) demonstrated that nocturnal surveys are the most efficacious method to determine the presence of adult and subadult California red-legged frogs, and they supported the more recent USFWS protocol (USFWS 2005) that requires both diurnal and nocturnal surveys to evaluate the presence of California red-legged frogs. Therefore, surveys that do not follow the most recent USFWS protocol are considered inadequate in determining that red-legged frogs would likely be absent from a site.

Two specimens at the Museum of Vertebrate Zoology, University of California at Berkeley, are very near, or within, the Tahoe National Forest. One of the historic locations is northeast of the town of Dutch Flat (T15N R10E, elevation 3200 feet). California red-legged frogs were collected at this site in 1916 and 1939. It is unknown if the location of this sighting was on private or public land. This site is approximately 3.3 miles from the nearest treatment unit proposed in this Project. The other historic sighting is near Michigan Bluff at Byrds Valley (T14N R11E, elevation 3200 feet). This record is from 1964 (J. Dixon). This site is on private land. A survey (1997) of this site by USGS NBS biologist noted that there is very little water available and that there was no suitable breeding habitat for frogs at this location.

Prior to the flooding of New Bullards Bar Reservoir several wetlands existed that could have supported California red-legged frogs. The finding (Sept. 15, 2000) of frogs in Little Oregon Creek (a tributary to New Bullards Bar Reservoir) on the Plumas National Forest could represent a remnant population from the wetlands that are now gone. This population is approximately 1 mile from New Bullards Bar Reservoir, and it is 13 miles from this Project boundary. Also to the north of Tahoe National Forest there is a California red-legged frog population at a pond on a private parcel in the French Creek drainage within Butte County (Barry 1999). This population is more than 10 miles from the Tahoe National Forest.

One mile south of the Tahoe National Forest, there is one known occurrence of this species on the Eldorado National Forest. On June 18, 2001, one female was detected in a pond on Ralston Ridge on the power line transmission corridor. The pond was dry several weeks later and dispersal of this individual remains unknown. In 2003 a population of California red-legged frogs was found on private land in a permanent pond on an ephemeral tributary to the South Yuba River. This site is near the Rock Creek watershed at approximately 3,000 feet in elevation.

In 2006, a red-legged frog site was discovered in the vicinity of Michigan Bluff on private land, near the town of Foresthill. Approximately 50 adults were observed in July 2006 inhabiting historic mine tailing ponds, at approximately 3,335 feet elevation. This occurrence is located just east of a historic occurrence reported from prior to 1951. In 2015 over 200 adult California red-legged frogs were observed at these ponds inside the Big Gun Conservation Bank (M. Young, personal communication, 2015). This population is considered to be the largest known Sierran population to date.

There is one perennial creek crossing associated with this project at 5400 feet in elevation, which is above the elevational range of this species. There is no suitable habitat associated with any of the proposed actions.

B. California Red-Legged Frog: Effects of the Proposed Action and Alternatives including Project Design Standards

Because this project is either outside of suitable habitat, or above the elevational range of the species, there are no direct, indirect, or cumulative effects that would occur.

C. California Red-Legged Frog: Conclusion and Determination

It is my determination that this project will have no effect on the California red-legged frog or its designated critical habitat.

LAHONTAN CUTTHROAT TROUT

Status: USFWS Threatened

A. Lahontan Cutthroat Trout: Existing Environment

The Lahontan cutthroat trout (LCT; Oncorhynchus clarkii henshawi) was listed by the USFWS as an endangered species in 1970 (USFWS 1970; 35 FR 13520). The listing was reclassified to threatened status in 1975 to facilitate recovery and management efforts and authorize regulated angling (USDI Fish and Wildlife Service 1975; 40 FR 29864). Currently, no Critical Habitat has been designated for the Lahontan cutthroat trout (USFWS 1995).

The USFWS is in the process of revising the 1995 Recovery Plan for Lahontan Cutthroat Trout. As part of recovery efforts, a technical team has assembled to develop restoration and recovery actions for the Truckee River basin. A primary purpose of the team is to identify and prioritize actions for the improvement of ecosystem function to facilitate the restoration/recovery of LCT. The USFWS believes that the establishment of lacustrine populations in Pyramid Lake, and in Lake Tahoe is necessary for the recovery of LCT in the Western Geographic Management Unit (GMU).

The Truckee River Basin Recovery Implementation Team (TRRIT) has been assembled to develop restoration and recovery actions for the Truckee River Basin. The TRRIT has established recovery objectives for various reaches of the Truckee River and its tributaries. Important recovery areas that the TRRIT has initially identified as having immediate potential include: Independence Creek, upstream of Independence Lake; Pole Creek; Hunter Creek; Donner Creek; Perazzo Creek; Prosser Creek; and the Truckee River from its confluence with Donner Creek to the State line; Upper Truckee River; Truckee River from Tahoe Dam to Donner Creek; and, Independence Creek downstream from Independence Lake to the Little Truckee River. The TRRIT has identified Macklin and East Fork Creeks and an unnamed tributary to the East Fork Creek in the Yuba River system as necessary for recovery of LCT because they contain remnants of indigenous Truckee River Basin strains.

In addition the TRRIT has drafted a Short-term Action Plan for LCT in the Truckee river Basin (USFWS 2003). This short-term (5 year) action plan includes a description of the elements needed for recovery, goals and objectives, timeline and priorities, actions needed and stakeholder participation plan.

Lahontan cutthroat trout is an inland subspecies (one of 14 recognized subspecies of cutthroat trout in the western United States) of cutthroat trout endemic to the Lahontan Basin of northern Nevada, eastern California, and southern Oregon. In northern California and western Nevada, LCT trout were thought to occupy approximately 656 miles of the Truckee River watershed, 400 miles of the Carson River watershed, and 570 miles of the Walker River watershed (USFWS

2009). Lahontan cutthroat trout historically occurred in Tahoe, Cascade, Fallen Leaf, Upper Twin, Lower Twin, Pyramid, Winnemucca, Summit, Donner, Walker, and Independence Lakes (Moyle 1976, Gerstung 1988). At the turn of the century, Lake Tahoe and Pyramid Lake supported commercial and sport fisheries for LCT. Self-sustaining populations of LCT trout are now extirpated from these lakes with the exception of Independence and Summit lakes (Behnke 1992). The Pyramid Lake LCT fishery is sustained by hatchery stocking (Somer 1998). Lahontan cutthroat trout has been extirpated from most of the western portion of its range in the Truckee, Carson, and Walker river basins, and from much of its historic range in the Humboldt basin (Gerstung 1988, Coffin 1988, USFWS 2009). Existing self-sustaining stream habitat in the Truckee, Carson and Walker river basins totals approximately 57 miles in headwater streams of northern California (USFWS 2009). Many of the stream populations occupy isolated segments of larger river systems with no opportunity for natural recolonization. Due to the fragmented, isolated nature of lake and stream populations, LCT may be at a high risk for extinction (USFWS 1995, Somer 1998, USFWS 2009, Moyle *et al.* 2011).

The severe decline in range and numbers of LCT attributed to a number of factors including hybridization and competition with introduced trout species; alteration of stream channels and morphology; loss of spawning habitat due to pollution and sediment inputs from logging, mining, grazing and urbanization; migration blockage due to dams; reduction of lake levels and concentrated chemical components in natural lakes; loss of habitat due to channelization; de-watering due to irrigation and urban demands; and overfishing (Gerstung 1986 & 1988, Coffin 1988, USFWS 1995, USFWS 2009).

The life history of LCT resembles that of other subspecies of interior cutthroat trout. Lacustrine LCT matures at 3-5 years of age (Gerstung 1988). In stream environments, males frequently mature at age 2, and females mature at age 3 (Coffin 1981). Lahontan cutthroat trout are obligatory stream spawners, with spawning occurring during spring months (generally April-July) depending on streams flow and water temperatures. Spawning LCT prefer gravel sizes ranging from 6 to 50 mm in diameter and water velocities ranging from 4-6 cm/s (Gerstung 1988). Spawning gravels must be clean and well oxygenated. Fecundity of 600 to 8,000 eggs per female has been reported for lacustrine populations (Calhoun 1942, Lea 1968, Cowan 1983, Sigler et al. 1983). However, fluvial females from small Nevada streams only had 100 to 300 eggs per fish (Cowan 1983). Fecundity and egg size are positively correlated with length, weight, and age (Sigler et al. 1983). Water temperatures of less than 13o C and intragravel dissolved oxygen levels in excess of 5 mg/l are required during the April through mid-August egg incubation period (USFWS 1995). Lahontan cutthroat trout eggs generally hatch in 4 to 6 weeks, depending on water temperature, and fry emerge 13 days later (Calhoun 1942, Lea 1968, Rankel 1976). Fry require habitats that include riffles, glides, and small pools. Fry prefer water depths of 6-43 centimeters and water velocities of less than 9 cm/s. Fry movement is density-dependent and correlated with fall and winter freshets (Johnson et al. 1983). Some fluvial adapted fish remain for 1 or 2 years in nursery streams before emigrating in the spring (Rankel 1976, Johnson et al. 1983, Coffin 1983). Growth rate is variable with faster growth occurring in larger, warmer waters and particularly where forage fish are utilized (Sigler et al. 1983). Growth rates for stream dwelling LCT are fairly slow (USFWS 1995).). Lahontan cutthroat trout may live 5–9 years in lake environments (Lea 1968, p. 26; Rankel 1976, p. 29; Rissler *et al*. 2006, p. 22) while stream dwelling LCT's are generally less than 6 years of age (Ray *et al*. 2007, pp. 39–60). Lahontan cutthroat trout may live 5–9 years in lake environments (Lea 1968, p. 26; Rankel 1976, p. 29; Rissler *et al*. 2006, p. 22) while stream dwellings LCT are generally less than 6 years of age (Ray *et al*. 2007, pp. 39–60).

Optimal habitat is characterized by 1:1 pool-riffle ratios; well vegetated, stable streambanks; over 25% cover; and relatively silt free rocky substrates (Hickman & Raleigh 1982). Lahontan cutthroat trout inhabit areas with overhanging banks, vegetation, or woody debris. In-stream cover (brush, aquatic vegetation, and rocks) is particularly important for juveniles (Sigler & Sigler 1987; Gerstung 1988). Lahontan cutthroat trout are unique since they can tolerate much higher alkalinities than other trout species (Koch et al. 1979). Adults can tolerate temperatures exceeding 27o C for short periods of time and seem to survive daily temperature fluctuations of 14-20o C (Coffin 1983; French & Curran 1991). Lahontan cutthroat trout do best in waters with average maximum water temperatures of less than 22o C and average water temperatures of 13o C. Stomach analysis of fluvial LCT showed that they are opportunistic feeders whose diets consist of organisms (typically insects) most commonly found in drift (Moyle 1976). In lakes, small Lahontan cutthroat trout feed largely on insects and zooplankton (Calhoun 1942, McAfee 1966, Lea 1968) and large LCT (>500 mm) feed on fish (Sigler *et al.* 1983).

Lahontan cutthroat trout evolved in the absence of other trout species and do not compete well for food and habitat. In stream environments within the western portion of the Lahontan drainage, LCT have seldom been able to co-exist with non-native trout for longer than a decade. Lahontan cutthroat trout, particularly those within the western portion of the Lahontan Basin, also hybridize with nonnative rainbow trout (Behnke 1979). The USFWS states that nonnative fish, particularly nonnative salmonids, are the primary threat to LCT (USFWS 2009).

Potential risk factors to the LCT include the immediate loss of individual fish and loss of specific habitat features such as undercut banks used for cover, increases in sedimentation leading to changes in spawning bed capacity, and the loss of riparian vegetation necessary to maintain adequate temperature regime (SNFPA 2001).

Conservation recommendations (USDI Fish and Wildlife Service 2001) that may be applicable to Tahoe National Forest management activities include:

1. Assist the USFWS in implementing recovery actions identified within Recovery Plan for the LCT, including:

    - Working with the USFWS and other agencies in developing Recovery and Implementation Plans for the Truckee River System.
    - Work with the USFWS, the CDFW to develop and implement a non-native, predator control program within occupied habitats for these species.
    - Educate the public on the adverse impacts of non-native predators on listed species.

2. Maintain at least 90 percent of natural streambank stability at the end of the authorized grazing season in areas that are occupied by LCT or LCT habitat within Critical Aquatic Refuges (CARs). This means that no more than 10 percent of the natural streambank

stability could be altered by activities such as, but not limited to: livestock trampling, chiseling and sloughing, OHV use, stream crossings, and recreational use.

3. At least 80 percent of natural streambank stability should be maintained at the end of the authorized grazing season in areas that are unoccupied by LCT but are potential habitat, within historic range. This should apply to all streams in the action area within the Truckee watershed. This means that no more than 20 percent of the natural streambank stability in these watersheds could be altered by factors such as, but not limited to: livestock trampling, chiseling and sloughing, OHV use, stream crossings, and recreational use.

4. Where unoccupied LCT streams are identified by the Service as egg incubation sites, reintroduction sites, or are identified as necessary to recover LCT, the FS should implement measures described in Measures 2 or 3 above, during the following grazing or use season for these identified unoccupied streams.

5. To aid recovery of threatened and endangered species and improve habitat conditions to allow expansion of existing populations, the FS should develop long-term allotment management plans (AMPs) for allotments necessary for the recovery of LCT. Basic features of the AMPs should include: 1) Appropriate combinations of pasture rest, grazing intensity, rotation and timing of livestock use; 2) exclosure fencing or riparian pasture management to allow continual and timely improvements in the condition of uplands and riparian vegetation and achievement of desired future condition; 3) establish monitoring programs to document changes in riparian and upland vegetation and stream habitat condition; and 4) use of alternate watering systems.

6. Dissolved oxygen, temperature, pH, specific conductance, nitrates, ammonia, total phosphorous, and total and/or fecal coliform bacteria should be regularly monitored in LCT streams to ensure adequate protection of water quality. At a minimum, sampling should occur: 1) prior to the seasonal introduction of livestock to the allotment, and 2) near the point in time when livestock are removed from the allotment. If monitoring demonstrates that constituent concentrations exceed standards for designated beneficial uses, a water quality management plan should be developed and implemented. If livestock contribute to constituent concentrations that exceed the standards, livestock should be removed from the stream and associated riparian area if compliance with water quality standards is not achieved within 2 years of implementation of the water quality management plan.

7. Designate all areas currently occupied by LCT as CARs.

Within Tahoe National Forest, recovery populations of LCT occur in one lake and five streams. Tahoe National Forest has designated the lake (Independence Lake) and the stream flowing into it (Upper Independence Creek) as a CAR. Management decisions and actions in a CAR should reflect the unique and important nature of the aquatic and riparian resources in these areas. Periodically CDFW surveys streams by electroshocker to determine LCT population trends. All populations in Tahoe National Forest have been stable and vary in numbers at carrying capacity for the habitat in the streams, However, the populations are small and may not be large enough to support genetic diversity in the long term, and may be at risk for genetic drift and bottlenecks (Somer, CDFW, personal communication, 2014).

Lahontan cutthroat trout do not range into the project area.

B. Lahontan Cutthroat Trout: Effects of the Proposed Action and Alternatives including Project Design Standards

Because this project occurs outside of the range of this species, there are no direct, indirect, or cumulative effects that would occur.

C. Lahontan Cutthroat Trout: Conclusion and Determination

It is my determination that this project will have no effect on the Lahontan cutthroat trout or its designated critical habitat.

WESTERN POND TURTLE

Status: USFS R5 Sensitive

A. Western Pond Turtle: Existing Environment

The western pond turtle (*Emys marmorata*) is found on the west coast of North America. Historically it was found from as far north as British Columbia, Canada to as far south as Baja California mostly west of the Cascade-Sierra crest (Lovich and Meyer 2002). Fossil fragments have been found east of the current range indicating that the species was once more widespread (Buskirk 2002). Disjunct populations have been documented in the Truckee, Humboldt and Carson Rivers in Nevada, Puget Sound in Washington, and the Columbia Gorge on the border of Oregon and Washington. It is currently unclear if these are relictual or introduced populations (Lovich and Meyer 2002). Modern distribution is limited to parts of Washington, Oregon, California and northern Baja California (Buskirk 2002). Western pond turtles are the only native aquatic turtle in California and southern Oregon, in the northern part of its range it coexists with only the western painted turtle (Chrysemys picta bellii) (Germano and Rathbun 2008). On Region 5 lands this turtle can be found on all National Forests, except the Inyo and Lake Tahoe Basin.

The Society for the Study of Amphibians and Reptiles no longer recognizes subspecies for the western pond turtle. Presumably this is based on recent genetic work that indicates that the recognized subspecies were not geographically or genetically correct, and the currently recognized species likely represents as many as four cryptic species. However, the study that identified the four distinct clades of pond turtle did not elevate any to species status as the authors wanted to wait until further molecular work was undertaken. The two former subspecies were the northwestern pond turtle (*Emys marmorata marmorata*) and the southwestern pond turtle (*Emys marmorata pallida*) with a subspecies split along the transverse mountain range in southern California (Spinks and Shaffer 2005).

Abundance has been well studied in this species. In some stream habitats densities can exceed 1,000 turtles per hectare. In Oregon, small ponds can hold over 500 turtles per hectare. These densities represent extremes with typical densities ranging from 23 to 214 turtles per hectare throughout most of the range (Lovich and Meyer 2002). Capture rates at one site in southern California were ca. 2 to 2.6 turtles per trap night (Germano 2010). These density estimates are likely accurate for populations on National Forest System lands where habitat is suitable.

The western pond turtle inhabits a Mediterranean climate defined by mild, wet winters and long hot, dry summers. In the northern portion of its range winters are colder with more rainfall than in southern areas (Germano and Rathbun 2008). Aquatic habitats include lakes, natural ponds, rivers, oxbows, permanent streams, ephemeral streams, marshes, freshwater and brackish estuaries and vernal pools. Additionally, these turtles will utilize man-made waterways including drainage ditches, canals, reservoirs, mill ponds, ornamental ponds, stock ponds, abandoned gravel pits, and sewage treatment plants (Buskirk 2002). Turtles captured at waste-water treatment plants grew quickly, had successful recruitment and produced large clutches (Germano 2010). Turtles favor areas with offshore basking sites including floating logs, snags, protruding rocks, emergent vegetation and overhanging tree boughs, but also will utilize steep and/or vegetated shores. Hatchlings additionally require shallow, eutrophic, warm areas which are typically at the margins of natural waterways (Buskirk 2002). Terrestrial habitats are less well understood. In southern California animals spend only one to two months in terrestrial habitats while animals in the northern portions of the range can be terrestrial for up to eight months (Lovich and Meyer 2002). Animals have been documented to overwinter under litter or buried in soil in areas with dense understories consisting of vegetation such as blackberry, poison oak and stinging nettle which reduces the likelihood of predation (Davis 1998).

Western pond turtles are generalist omnivores and have been documented to eat a wide variety of prey. Prey items include larval insects, midges, beetles, filamentous green algae, tule and cattail roots, water lily pods, and alder catkins (Germano 2010). Filamentous algae is considered to be an important food source for females after egg laying (Buskirk 2002). Additionally, animals will opportunistically feed on other items such as floating duck carcasses, ducklings (pers. obs.) and dog food in backyards while on walkabouts (Buskirk 2002).

Turtles move upland at different times across the range of this species. Animals can move upland as early as September, but typically move following the first winter storm in November or December. Not all animals move upland, some move to nearby ponds for the winter (Davis 1998). Animals have been observed moving underneath ice in ponds and potentially congregate in shallow areas (Buskirk 2002). Upland animals remain somewhat active throughout the winter and can be observed basking on warm winter days (Davis 1998). Upland movements for both overwintering and reproduction typically occur in the afternoon and evenings. Walkabouts to scout for nest sites can be completed within one day or they can last up to four days (Crump 2001). Home ranges differ between males and females with male home ranges averaging 0.976 hectares and females averaging 0.248 hectares. Although western pond turtles are likely not territorial, disputes over basking sites are commonly observed (Buskirk 2002).

Local climatic and water level variations can alter the timing of nesting in this species (Crump 2001). The nesting season is from late April through mid-July at low elevation, and June through August at higher elevations (Scott et al. 2008). Although some females can reproduce with a carapace length as small as 111 mm, 120 mm is the minimum reproductive size in most areas with most gravid females being 140 mm or larger (Scott et al. 2008). Animals of this size are often at least seven years old in southern areas and eight to twelve years old in northern areas. Western pond turtles have an average life expectancy of approximately forty years if they survive to adulthood (Buskirk 2002).

Some western pond turtles have shown nest site fidelity. Four of five detected nesting areas in one study area had instances of nest side fidelity. It is likely that nest site fidelity is common, and sites are changed only after a negative encounter during either a walkabout or while forming a nest at a particular site (Crump 2001). Most females nest within 50 meters of water; however some females nest upwards of 400 meters away from water (Lovich and Meyer 2002). It is believed that in coastal populations nesting occurs far from water in order to protect overwintering hatchlings from being injured during winter floods (Lovich and Meyer 2002). Mean clutch size ranges from 4.5 +/- 0.25 on the Santa Rosa Plateau to 7.3 +/- 1.18 in southern Oregon. More research is needed to determine if clutch size varies with latitude (Germano and Rathbun 2008). Average annual egg production for 39 animals in southern California was 7.2 +/- 3.9 eggs. This number did not vary statistically among females of differing carapace length or among different streams and in many cases represented two clutches per female. Clutch size varies significantly among drainages; however it does not differ significantly across years or within individual drainages. When double clutching occurs, the first clutch typically contains more eggs than the second clutch (Scott et al. 2008). Several head-starting programs claim that temperature dependent sex determination is utilized by western pond turtles, but they have not published evidence proving this (Buskirk 2002).

Hatchlings in the Mojave River population overwinter in the nest and emerge as early as March of the following year (Lovich and Meyer 2002). However, most hatchlings in southern California emerge in late fall of the year they were laid. Northern animals typically emerge the following spring. Delayed emergence can be caused by soil structure where sandy soil results in earlier emergence (Crump 2001). Microhabitat use, behavior and diet differ between juvenile and adult western pond turtles (Lovich and Meyer 2002). Little is known about the specific requirements of hatchling turtles as they are cryptic and are rarely represented in population assessments of many species including those with known stable populations (Germano and Rathbun 2008). Growth and maturation in western pond turtles is heavily influenced by ambient air and water temperatures and basking behaviors which include aerial basking, and cryptic behaviors such as burying in warm sand or lying in warm algal mats (Germano and Rathbun 2008). Sites with cold water require turtles to bask more causing average body size to be smaller compared to sites with warmer water. Areas which have higher invertebrate densities typically classified as having organic mud bottom substrates yield larger turtles (Lubcke and Wilson 2007).

Western pond turtles have significantly declined in number with many populations representing less than 10% of the historical population. In California alone there has been a loss of 80-85% of western pond turtles since the 1850's. The Puget Sound population in Washington, which encompassed the type location for this species, as well as British Columbia populations has been considered extirpated since at least the 1970s. 98% of the population is gone in Oregon's Willamette Valley, 95-99.9% of the population in the San Joaquin Valley is gone and most of the

Nevada populations have disappeared. In southern California there are only 12 known viable populations (>25 adult animals) between Los Angeles County and the Mexican border (Buskirk 2002).

The major threat to this species is habitat loss or degradation. Most of the historical habitat for this species has been permanently lost as a result of development for human occupancy. Riparian and wetland habitats are cleared for agriculture use, destroyed by cattle, channelized and stripped of vegetation, or invaded by the saltcedar shrub which destroys water quality, alters stream structure and dries streams. Ground water pumping lowers water tables and further stresses riparian plant communities. Gold and gravel mining can both directly destroy habitat as well as introduce toxins through toxic spills and illegal dumping of chemicals (Buskirk 2002; Lovich and Meyer 2002).

Additional human-caused threats further jeopardize population viability. Cattle grazing can destroy riparian habitat, trample and kill turtles and nests, and cattle waste can pollute waterways. Western pond turtles, especially gravid females, are easily killed on roadways by direct impact with vehicles. Historically animals were also collected for the pet trade with hundreds of animals from a single site being exported to Europe in the 1960's. Although collection and sale of western pond turtles have been banned for many years, animals are still listed for sale in the eastern United States. Animals were collected for food in great numbers from the mid-19th century to the 1930s when animals first started to become scarce. Modern watercourse recreation also impacts these turtles. Recreation which interferes with basking or causes direct injury or mortality include high-speed boating, water skiing, jet skiing and fishing where animals may be directly caught or killed because they are viewed as competition (Buskirk 2002).

Disease poses a notable threat to western pond turtles, as has been seen in Washington. A die-off in 1990 was attributed to a syndrome similar to an upper-respiratory disease. Several years later, as part of a head-starting program, several animals were found dead with no apparent cause of death (Vander Haegen et al. 2009). Animals from a wastewater treatment pond in California were found to be less healthy in both the short and long term compared to animals in a natural habitat despite being larger in size. Although larger, these animals had more chronic stress in the form of more interactions with humans and invasive species, increased water pollution and greater exposure to water-borne diseases (Polo-Cavia et al. 2010). Dehydration also poses a threat to turtles under a year old which likely makes these animals more susceptible to disease (Vander Haegen et al. 2009).

In addition to threats that affect entire populations, many populations are failing as a result of extremely high juvenile mortality. While adults may have annual survival rates of 95-97%, nests, juveniles and sub-adults have extremely high mortality rates (Vander Haegen et al 2009). Nest destruction by raccoons can approach or reach 100% of nests at many Oregon nest sites (Buskirk 2002). Nests are also destroyed when exposed to too much moisture or are crushed by cattle or machines. There are many predators of hatchling turtles, including two very successful non-native predators- large-mouth bass and bullfrogs. Subadult mortality can be as high as 85-90% annually for animals under 4 years old, however head-started subadults had mortalities as low as 10% when carapace length was greater than 90mm. Natural predators that have been documented to take sub-adult turtles include: raccoons, coyotes, black bears and western river otters with most predations occurring while the animal was terrestrial (Vander Haegen et al. 2009). Adults face less predation risk. A study documented one predation of an adult turtle by a loon, and only 3 of 196 turtles had evidence of predation attempts such as shell or limb damage (Davis 1998).

Few turtle specific surveys have been conducted in the Tahoe National Forest. Primarily, northwestern pond turtle observations have been made during aquatic surveys or other forest activity surveys. California Academy of Sciences, San Francisco, has conducted herpetological surveys including areas likely to provide habitat for northwestern pond turtles (1997, 1998, 1999). Western pond turtles have been observed at approximately 30 locations within the Tahoe National Forest boundary. Tahoe National Forest's recorded sightings are from the Yuba River drainage and the American River or it's tributaries. Most of the observations have been associated with pond habitats, although several observations were of turtles walking distant from an aquatic habitat (e.g. turtle walking across a road).

There is one perennial creek crossing associated with this project,that occurs at 5400 feet in elevation, which is above the elevational range of this species.

B. Western Pond Turtle: Effects of the Proposed Action and Alternatives including Project Design Standards

This project either occurs outside of habitat, or where habitat may be present, it is above the elevational range of this species. Therefore, there are no direct, indirect, or cumulative effects that would occur.

C. Western Pond Turtle: Conclusion and Determination

It is my determination that implementation of Alternative **__** will not affect the northwestern pond turtle.

FOOTHILL YELLOW-LEGGED FROG

Status: USFS R5 Sensitive

A. Foothill Yellow-Legged Frog: Existing Environment

The foothill yellow-legged frog (*Rana boylii*) is listed as Sensitive on the Region 5 Forester's Sensitive Species List (USDA Forest Service 1998) and is considered a Species of Concern by the state of California. Standards and guidelines for Riparian Conservation Areas provide management direction for foothill yellow-legged frog and are described in the SNFPA ROD (January 2004).

Foothill yellow-legged frogs (Rana boylii) have suffered significant population declines across the majority of the known range. Historically this frog was found across most of southwestern Oregon west of the Cascades Mountains crest south through California to Baja California (Fellers 2005; Jennings and Hayes 1994). Specimens collected from the Sierra San Pedro Martir of Baja California in 1961 were lost in transit and represented a population almost 300 miles south of the nearest known population (Loomis 1965). The foothill yellow-legged frog is found in most of northern California west of the Cascade Mountains crest, in the Coast Ranges from the California-Oregon border south to the Transverse Mountains in Los Angeles County and along the western slope of the Sierra Nevada Mountains south to Kern County. Isolated populations have been reported from the San Joaquin Valley and the mountains in Los Angeles County. This frog can be found from near sea level to 1940m (6370 ft) where habitat is suitable (Morey 2000). Within Region 5 this frog is found on, or could occur on, all national forests except for the Cleveland, Inyo, Modoc, and Lake Tahoe Basin National Forests.

Populations of foothill yellow-legged frogs in the Pacific Northwest are considered to be the most stable with approximately 40% of streams occupied, 30% are occupied in the Casade Mountains, 30% in the south Coast Range south of San Francisco and 12% in the Sierra Nevada foothills (Fellers 2005). Populations in and south of the Tehachapi Mountains have probably been extirpated (Santos-Barrera et al. 2004). The last verifiable record from this area is a series of animals which were collected in 1970 however unverifiable observations occurred through the late 1970's (Jennings and Hayes 1994). Any remaining populations in Mexico are protected by Mexican law under the "Special Protection" category (Santos-Barrera et al. 2004). While there are no recognized subspecies of foothill yellow-legged frogs, recent genetic studies indicate that there is a genetic break along the transverse mountains (Lind et al. 2011). Although there are numerous occupied streams, only 30 of the 213 known populations in California have populations of at least 20 individual adults. These frogs are most numerous in the northern coast range with six populations of at least 100 adults and an additional nine populations of at least 50 adults (Fellers 2005). An assessment by Lind (2005) found that foothill yellow-legged frogs were not present at 51% of their historically occupied sites in the Sierra Nevada.

Foothill yellow-legged frogs are found in partially shaded, rocky, perennial and ephemeral streams, rivers, and wet terrestrial habitats (Hays et al. 2016). They are found in a variety of habitats including: valley-foothill hardwood, valley-foothill hardwood-conifer, valley-foothill riparian, ponderosa pine, mixed conifer, coastal scrub, mixed chaparral and wet meadows and appear to be highly dependent on free water for all life stages (Morey 2000).

Breeding habitat selection ranges from smill tributaries to large rivers (Lind 2005). In large rivers breeding sites are commonly point bars or depositional sites near tributary confluences (Kupferberg 1996). Smaller stream breeding habitat is typically classified as a stream with riffles containing cobble-sized or larger rocks as substrate (Morey 2000). These streams are further defined by having low-water velocities near tributary confluences in shallow reaches and are wider and shallower than non-breeding sites, have emergent rocks and are typically asymmetrical with cobble or small boulder bars (Wheeler and Welsh 2008; Kupferberg 1996). Egg attachment sites are usually cobbles or boulders, but frogs may sometimes utilize bedrock or vegetation. These sites are often on the lee side of rocks or beneath overhangs such that the site has a narrow range of low-water velocity. Coarse sediment enables frogs to choose the best oviposition site to shield egg masses from high-flows. The reproductive strategy of the foothill yellow-legged frog is well suited to rivers with predictable winter flooding and summer droughts (Kupferberg 1996).

As tadpoles stream velocities remain important and high flows can flush tadpoles down stream (Kupferberg et al. 2008). Tadpoles remain in the same general habitat as the breeding site, which has low flows and refugia from high velocity events (Bondi et al. 2013, Kupferberg et al. 2008).

The habitat preferences of non-breeding adult foothill yellow-legged frogs include riparian and aquatic habitats. Selected habitat consists of shallow, low gradient streams with coarse substrate (Morey 2000) Adults may migrate up tributaries consisting of large-sized boulders and bedrock to utilize the cooler air and water temperatures, and to avoid predators and high water flows (Leidy et al. 2009).

Overwintering habitat as summarized in the Foothill Yellow-Legged Frog Conservation Assessment (Hayes 2016):

> "Overwintering is the least understood aspect of foothill yellow-legged frog habitat use. Van Wagner (1996) observed postmetamorphic frogs overwintering in velocity-protected areas of the channel (e.g., on the lee side of sedge [Carex spp.] tussocks). He found that adult frogs typically used root wads, woody debris, undercut banks, and large boulders adjacent to pools, whereas juvenile frogs were usually found in hollows at the stream edge created by sedges partially springing back from being pushed over during high flows. Van Wagner's (1996) observations occurred in a relatively small stream, where enough protection from scour or bedload movement may exist in selected in-channel locations during high discharge. Recent metamorphs have been caught in pitfall traps moving upland away from mainstem channels (Twitty et al. 1967); observed in wet ditches along dirt roads (Kupferberg, 2012), and in caves and tunnels (Devine Tarbell & Associates, Inc. and Stillwater Sciences 2005). In larger streams, frogs may use protected terrestrial sites that avoid the scour risk entirely. For example, in the South Santiam River in Oregon (a 6th-order channel), juveniles occupy seeps above the typical winter high-flow waterline (Rombough 2006b); whether adults overwinter terrestrially at this site is unknown. Adult frogs observed moving upland along the Trinity River during fall rains may be movement into terrestrial sites (M. Hayes, personal observation, 1994; Jennings 1990), or to lateral tributaries for overwintering, where scour risk may be considerably lower (Kupferberg 1996a). Frogs have been observed in terrestrial habitats far from streams in the South Fork Eel and Mattole watersheds of the Coast Ranges in late fall."

Wheeler and Welsh (2008) found that approximately 68% of adult male foothill yellow-legged frogs in their study were site faithful. These animals had an average breeding home range of $0.58 m^2$ and home range size was directly linked with the frequency of aggressive behavior and calling activity. Males were not actively guarding future oviposition sites, but were guarding a specific, but generalized, patch of habitat within the breeding site (Wheeler and Welsh 2008).

Larval foothill yellow-legged frogs primarily consume algae and will preferentially graze on epiphytic diatoms as this food item allows them to grow more rapidly (Jennings and Hayes 1994). Post-metamorphs are generalists that eat mostly terrestrial insects but also aquatic insects (Hayes et al. 2016). Adult diet is thought to include: flies, moths, hornets, ants, beetles, grasshoppers, water striders and snails with a terrestrial arthropod composition of 87.5% insects and 12.6% arachnids (Fellers 2005).

Breeding can occur as early as March but mainly occurs in May to early June and varies in length with an average duration of 49.5 days between first and last egg depositions (Hayes 2016, Wheeler and Welsh 2008; Kupferberg 1996). Breeding occurs earlier in low-base flow years and begins when stream flow is at or below 0.6m/sec and between 0.04 and 0.17 m/sec at the microhabitat scale (Wheeler and Welsh 2008). Eggs are typically laid in shallow areas ranging from 4 to 43 cm at varying distances from shore. When base-flow is low frogs will oviposit further from shore (Kupferberg 1996; Lind et al. 1996). Prior to egg deposition but while in amplexus, females will scrape potential attachment sites with their hind-feet in order to remove any debris and make egg adhesion stronger. This reduces the likelihood of the clutch being detached by a change in water velocity (Rombough and Hayes 2005). Females lay a single annual clutch of between 300 and 2,000 eggs (Jennings and Hayes 1994; Kupferberg 1996). Reproductive output is typically 18.8 +/- 1.9 clutches per breeding site (Kupferberg 1996). The critical thermal maximum for embryos is approximately 26C, however eggs are typically found from 9 to 21.5C (Jennings and Hayes 1994; Kupferberg 1996). Incubation lasts approximately two weeks (5 – 37 days) depending on water temperature and position within the clutch. Eggs near the attachment point and eggs in the center of the clutch typically hatch later than eggs on the periphery of the clutch (Kupferberg 1996; Fellers 2005). After hatching, tadpoles move away from the egg mass. As with egg development, larval development is temperature dependent with metamorphosis typically occurring 2-4 months after hatching depending on the temperature and quality of diet (Catenazzi and Kupferberg 2013, Kupferberg et al. 2011) with no documented overwintering of larvae. Foothill yellow-legged frogs metamorphose at a size of 1.4-1.7 cm in length. Reproductive maturity is thought to occur the second year after metamorphosis, but can occur as early as six months after metamorphosis. Longevity for this species is unknown (Fellers 2005).

High mortality in this species occurs during the egg and larval life stages. The main causes of mortality in eggs are hydrologic in nature. Eggs are usually killed by either desiccation or scour (Kupferberg 1996; Lind et al. 1996). Tadpole mortality can also occur as a result of irregular stream flows. The main critical velocity for tadpoles is 20cm/s but flows as low as 10cm/s can displace large tadpoles. This results in slower growth and development, greater exposure to predators and possible mortality. The seasonal pulses of high water flows used in many regulated rivers have a significant negative impact on recruitment for this species (Kupferberg et al. 2011). Variability in spring and summer flows was found to cause egg mass and tadpole mortality and decreases in river levels can cause stranding (Kupferberg 1996, Kupferberg et al. 2016, Wheeler et al. 2013).

Loss of genetic diversity due to habitat loss is a major threat to foothill yellow-legged frogs. Populations which are more than 10km apart are prone to genetic drift and barriers such as dams or habitat fragmentation may prevent dispersal between isolated populations (Dever 2007). Due to the isolation it is unlikely that extripated populations will recolonize (Hayes et al. 2016). In one study area, 94% of downstream bar habitat and potential breeding habitat was lost after the installation of a dam. The encroachment of riparian vegetation created stable sandy berms which caused the river to become narrower and deeper and thus unsuitable for use by foothill yellow-legged frogs (Lind et al. 1996).

The five major risk factors as designated in the Foothill yellow-legged frog Conservation Assessment (Hayes et al. 2016) are: water development and diversion, climate change, habitat loss (urbanization and fragmentation), introduced species, and mining.

In the Tahoe National Forest, foothill yellow-legged frog occurances have been recorded as far back as the late 1800's. Collection records document occurances mainly for the Yuba River, with a few occurances recorded in the American River System (Hayes et al. 2016). Surveys were conducted in cooperation with the USGS Biological Division, Pt. Reyes from 1997 through 2000. In addition, California Academy of Sciences, San Francisco, has conducted herpetological surveys including areas likely to provide habitat for mountain yellow-legged frogs (1997, 1998, and 1999). Foothill yellow-legged frog surveys were conducted at 46 sites in the treatment units and one reference location at the North Fork American River in 2007, by consultants for Placer County Water Agency. Individual frogs of different life stages were found in the mainstem and tributaries of the Middle Fork American River, North Fork American River, and the North Fork Middle Fork American River. Under the hydropower relicensing for the Yuba-Bear, Drum-Spaulding projects, foothill yellow-legged frog surveys were conducted in 2008, 2009, and 2010 within four accessible portions of the Middle Yuba and South Yuba Rivers, beginning at the Wolf Creek confluence at approximately 3,000 feet in elevation and downstream. Foothill yellow-legged frog breeding was confirmed at all survey sites and within tributaries at the four locations. Surveys conducted in 2016 have confirmed presence of Foothill yellow-legged frogs in Shritail Creek on the American River Ranger District. Amphibian occurrence is also documented during fish stream surveys and incidental to various other field activities and surveys.

This species is known to be present in the North Yuba River and its tributaries, with most occurrences below 3,000 feet in elevation. Only one perennial creek crossing is proposed across Rattlesnake Creek at 5400 feet in elevation. It is not likely that this species is present at this site.

B. Foothill Yellow-Legged Frog: Effects of the Proposed Action and Alternatives including Project Design Standards

Direct and Indirect: One re-route for the has a perennial creek crossing in Rattlesnake Creek at 5400 feet elevation, which is above the elevational range for this species. Therefore, there are no direct, indirect, or cumulative effects that would occur.

C. Foothill Yellow-Legged Frog: Conclusion and Determination

It is my determination that implementation of this project will not affect the foothill yellow-legged frog.

SIERRA NEVADA YELLOW-LEGGED FROG

Status: USFS R5 Sensitive, USFWS Endangered

A. Sierra Nevada Yellow-Legged Frog: Existing Environment

On January 10, 2003 (and as revised on June 25, 2007) the USFWS found that listing of the Sierra Nevada yellow-legged frog as threatened or endangered was warranted but precluded by higher priority actions and the species was listed as a Candidate (USFWS 2003; 68 FR 2283, and revised by USFWS 2007; 72 FR 34657). A separate disjunct population, the southern California DPS of the mountain yellow-legged, was listed as Endangered by the USFWS effective August 1, 2002 (USFWS 2006; 67 FR 44382), and critical habitat was designated for the southern California DPS on October 16, 2006 (USFWS 2006; 71 FR 54344). On April 29, 2014, the U.S. Fish and Wildlife Service published a final rule in the Federal Register to list the Sierra Nevada yellow-legged frogs and Northern DPS mountain yellow-legged frogs as endangered with extinction (USFWS 2014). The rule went into effect on June 30, 2014. A Critical habitat (CH) designation was proposed by the USFWS (2013), and was finalized as of August 26, 2016. The Forest Service will conduct a separate programmatic/appended BA to analyze potential effects to critical habitat for these species.

The mountain yellow-legged frog was once considered two subspecies of the *Rana boylii* group, with one of the subspecies in southern California which was disjunct from the one in the Sierra Nevada, and was later described as a single species, *Rana muscosa*. Genetic analysis conducted by Macey et al. (2001) indicated that there were at least four evolutionarily distinct units within *Rana muscosa*, and two major clades that diverged approximately 2.2 million years ago; one in the northern and central Sierra Nevada, and one in the southern Sierra Nevada and southern California. Recent genetic analysis combined with morphological and acoustic studies have described *Rana muscosa* as two separate species, *Rana muscosa* (mountain yellow-legged frog) and *Rana sierrae* (Sierra Nevada yellow-legged frog). Vredenburg et al. (2007) found no overlap in the ranges of the two species that they described, but their ranges come very close to each other in the southern Sierra Nevada, with *Rana sierrae* to the north and *Rana muscosa* to the south including the disjunct southern California population (Vredenburg et al. 2007).

Rana sierrae can be found on the El Dorado, Inyo, Lassen, Plumas, Sierra, Stanislaus, Tahoe and Lake Tahoe Basin National Forests. This species is found from around 4,500 feet to over 12,000 feet elevation, and inhabit ponds, lakes, and streams of sufficient depth for overwintering (Jennings and Hayes 1994). Yellow-legged frogs are highly aquatic, utilizing only the immediate bank and emergent rocks and logs. Their preferred aquatic habitat consists of stream or lakes with a gentle slope such that at the shore there is shallow warm water. Historically streams with a bank of less than 10 inches in vertical height with a moderately rocky, sparsely vegetated bank harbored the densest populations (Mullally and Cunningham 1956).

Critical habitat was defined by the USFWS (2016) using current and historic detections (SNYLF that have been confirmed since 1995) and modeling important habitat attributes (MaxEnt 3.3.3e) to produce the likelihood of frog occurance. Dr. Knapp's model (MaxEnt 3.3.3e) used ten environmental factors to determine likelihood of frog occurrence which include; elevation, max. elevation of unit watershed, slope, avg. annual temperature, average annual temperature, average temperature of the warmest month of the year, annual precipitation, precipitation during the driest month of the year, distance to water, and lake density. The USFWS defined primary habitat requirements below.

“Based on our current knowledge of the physical or biological features and habitat characteristics required to sustain the species’ life-history processes, we determine that the primary constituent elements specific to the Sierra Nevada yellow-legged frog and the northern DPS of the mountain yellow-legged frog are:

(1) *Aquatic habitat for breeding and rearing.* Habitat that consists of permanent water bodies, or those that are either hydrologically connected with, or close to, permanent water bodies, including, but not limited to, lakes, streams, rivers, tarns, perennial creeks (or permanent plunge pools within intermittent creeks), pools (such as a body of impounded water contained above a natural dam), and other forms of aquatic habitat. This habitat must:(a) For lakes, be of sufficient depth not to freeze solid (to the bottom) during the winter (no less than 1.7 m (5.6 ft), but generally greater than 2.5 m (8.2 ft), and optimally 5 m (16.4 ft) or deeper (unless some other refuge from freezing is available)). (b) Maintain a natural flow pattern, including periodic flooding, and have functional community dynamics in order to provide sufficient productivity and a prey base to support the growth and development of rearing tadpoles and metamorphs. (c) Be free of introduced predators. (d) Maintain water during the entire tadpole growth phase (a minimum of 2 years). During periods of drought, these breeding sites may not hold water long enough for individuals to complete metamorphosis, but they may still be considered essential breeding habitat if they provide sufficient habitat in most years to foster recruitment within the reproductive lifespan of individual adult frogs.(e) Contain: (i) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel, cobble, rock, and boulders (for basking and cover); (ii) Shallower microhabitat with solar exposure to warm lake areas and to foster primary productivity of the food web; (iii) Open gravel banks and rocks or other structures projecting above or just beneath the surface of the water for adult sunning posts; (iv) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, or rocks and vegetation to provide cover from predators; and (v) Sufficient food resources to provide for tadpole growth and development.

(2) *Aquatic nonbreeding habitat (including overwintering habitat).* This habitat may contain the same characteristics as aquatic breeding and rearing habitat (often at the same locale), and may include lakes, ponds, tarns, streams, rivers, creeks, plunge pools within intermittent creeks, seeps, and springs that may not hold water long enough for the species to complete its aquatic life cycle. This habitat provides for shelter, foraging, predator avoidance, and aquatic dispersal of juvenile and adult mountain yellow-legged frogs. Aquatic nonbreeding habitat contains: (a) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel, cobble, rock, and boulders (for basking and cover); (b) Open gravel banks and rocks projecting above or just beneath the surface of the water for adult sunning posts; (c) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, or rocks and vegetation to provide cover from predators; (d) Sufficient food resources to support juvenile and adult foraging; (e) Overwintering refugia, where thermal properties of the microhabitat protect hibernating life stages from winter freezing, such as crevices or holes within bedrock, in and near shore; and/or (f) Streams, stream reaches, or wet meadow habitats that can function as corridors for movement between aquatic habitats used as breeding or foraging sites.

(3) *Upland areas.*
(a) Upland areas adjacent to or surrounding breeding and nonbreeding aquatic habitat that provide area for feeding and movement by mountain yellow-legged frogs. (i) For stream habitats, this area extends 25 m (82 ft) from the bank or shoreline. (ii) In areas that contain riparian habitat and upland vegetation (for example, mixed conifer, ponderosa pine, montane conifer, and montane riparian woodlands), the canopy overstory should be sufficiently thin (generally not to exceed 85 percent) to allow sunlight to reach the aquatic habitat and thereby provide basking areas for the species. (iii) For areas between proximate (within 300 m (984 ft)) water bodies (typical of some high mountain lake habitats), the upland area extends from the bank or shoreline between such water bodies. (iv) Within mesic habitats such as lake and meadow systems, the entire area of physically contiguous or proximate habitat is suitable for dispersal and foraging. (b) Upland areas (catchments) adjacent to and surrounding both breeding and nonbreeding aquatic habitat that provide for the natural hydrologic regime (water quantity) of aquatic habitats. These upland areas should also allow for the maintenance of sufficient water quality to provide for the various life stages of the frog and its prey base." (USFWS 2016).

Rana sierrae primarily feed on aquatic and terrestrial invertebrates along the shoreline and on the water surface (Vredenburg et al. 2005), while larvae feed on benthic algae and detritus (Knapp et al. 2003). Pope and Matthews (2001) noted that seasonal movements appeared to be correlated to the abundance of tree frog larvae (Hyla regilla), a prey species of adult mountain yellow-legged frogs. Pope and Matthews (2002) found that abundance of tree frog larvae in a water body as a source of prey positively influenced the condition of mountain yellow-legged frogs, especially important leading into winter. Pope and Matthews (2002) also analyzed species occurrence data of lakes across the John Muir Wilderness and Kings Canyon National Park, and found that adult yellow-legged frogs were more abundant in lakes with other frog species than in lakes with no other frog species, and suggested this pattern was due to other frog species' larvae used as a food source.

All age classes (subadult and adult frogs, and larvae) overwinter underwater; in high elevations they are restricted to relatively deep lakes (over 5 feet deep) that do not freeze solid in winter (Knapp 1994, Knapp and Matthews 2000). Frogs (subadults and adults) hibernate underwater in winter; winterkill of subadults and adults may occur due to oxygen deprivation over winter under ice, while larvae are more resistant (Bradford 1983). Larvae require 2 to 4 years to metamorphose, and thus require water bodies which do not dry in summer (Knapp and Matthews 2000). At least some of the population overwinters in shallow lakes (<1.5m) that likely freeze to the bottom most years. These frogs likely avoid freezing by utilizing underwater crevices (Pope and Matthews 2001). Frogs utilize near shore ledges and crevices in fractured bedrock along the shoreline which are close to the water's surface (0.2 to 1m). These crevices are typically very narrow, but may open to larger areas deeper within the rock and often contain multiple individuals indicating that this species overwinters in aggregations. Both aggregations and the surrounding granite likely insulate individual animals from temperature extremes throughout the winter (Matthews and Pope 1999). Site fidelity is high for breeding, foraging and overwintering for this species (Matthews and Preisler 2010).

Breeding occurs soon after spring thaw, ranging from April at lower elevations to June or July in high elevations (Vredenburg et al. 2005). During spring thaw, frogs emerge to the surface to bask in the sun, or travel over ice and snow to other nearby bodies of water (Pope and Matthews 2001), while larvae seek warmer water near shore (after spring turnover in large bodies of water) (Bradford 1984). Sierra Nevada yellow-legged frogs lay their eggs in clusters submerged in shallow areas (Bradford 1983), under banks or attached to rocks, gravel, or vegetation (Vredenburg et al. 2005). The length of the larval stage depends on elevation; larvae require at least one year before metamorphosis to the adult stage, but most Sierra Nevada populations are composed of larvae in three size classes which may correspond to year classes (Vredenburg et al. 2005). Metamorphosis occurs in July or August (Vredenburg et al. 2005). The time required to reach reproductive maturity is believed to vary between 3 and 4 years after metamorphosis (Vredenburg et al. 2005), and adult survivorship is very high (Matthews and Pope 1999).

During summer, frogs and larvae seek the warmest thermal regimes throughout the day and night (Bradford 1984). Adults are rarely far from water, usually less than 1 meter and almost always on a wet substrate while basking, typically from sunrise into late morning (Bradford 1984). Bradford (1984) observed daily movements of adults corresponding to areas of warmer temperatures; in morning they basked in sun, were in water near shore from mid day until nightfall, and submerged in warmer deeper water for most of the night, usually under rocks or in crevices. Larvae exhibited similar selection for warmer temperatures throughout the day and night, as well as seasonally; they stay in deeper, warmer water below the thermocline until spring turnover, at which time they move to shallow water near the shoreline for the daytime and deeper, warmer water at night (Bradford 1984). Highest summer densities and overall total numbers are found in lakes lacking introduced fish, possessing high numbers of Pseudacris regilla tadpoles, more than 1 meter in depth and near-shore habitat with warm water temperatures (Pope and Matthews 2001).

In a relatively small basin (0.4 mi^2) with numerous small lakes and stream segments in Kings Canyon National Park, Matthews and Pope (1999) and Pope and Matthews (2001) observed seasonal movement patterns that coincided with changes in activity from overwintering habitat to breeding and feeding habitat and back again to overwintering habitat. Pope and Matthews (2001) observed frogs moving between nearby lakes over snow and ice during the spring thaw, and one frog was found wandering upslope about 200 feet from the basin water bodies on snow. Frogs were also observed moving overland in late summer to disperse to other nearby aquatic habitats, likely in response to reduced prey availability; some individuals moved overland for distances of at least 466 feet to other nearby aquatic habitats as summer progressed (Pope and Matthews 2001). A study by Finlay and Vredenburg (2007) in the Sixty Lakes Basin (approximately 12 mi2 study area) in Kings Canyon National Park where there are numerous water bodies in close proximity to each other suggests that when small lakes and ponds are used for breeding, frogs may leave these areas for other nearby aquatic habitats that lack mountain yellow-legged frog larvae. Matthews and Pope (1999) found that frogs tended to be relatively stationary in August when feeding appeared important and were often found in the open, then moved to overwintering locations in September, and were stationary by the end of October under ledges and in rock crevices and rarely in the open.

Once abundant in aquatic ecosystems of the mid to high elevation Sierra Nevada from southern Plumas County to southern Tulare County (Jennings and Hayes 1994), the mountain yellow-legged frog has undergone a range-wide decline in the Sierra Nevada (USFWS 2003). Over 90% of historically occupied sites in the Sierra Nevada are now unoccupied (Vredenburg et al. 2007).

The decline of mountain yellow-legged frogs in the Sierra Nevada has largely been attributed to the introduction of salmonid fishes during the last century (USFWS 2003). More recently, the disease chytridiomycosis has emerged as a significant threat to the species (Briggs et al. 2005, Oullet et al. 2005, Wake and Vredenburg 2008). Additional reasons for the mountain yellow-legged frog decline or contributing factors include airborne pesticides (Davidson et al. 2002, Davidson 2004, Davidson and Knapp 2007), loss of habitat, altered habitat, and grazing (USFWS 2003). Davidson and Knapp (2007) evaluated over 6800 sites in the southern Sierra Nevada comparing mountain yellow-legged frog occupancy with presence of introduced fish, habitat conditions, and predicted exposure to airborne pesticides from agricultural lands upwind in California's Central Valley, and found that airborne pesticides appeared to have a pronounced negative effect on mountain yellow-legged frog occupancy independent of the other factors examined.

Predators known to consume yellow-legged frogs include garter snakes (*Thamnophis spp.)* (Mullally and Cunningham 1956), Eared Grebes which prey on both tadpoles and small frogs (Fellers et al. 2007). In at least one instance an entire year's worth of metamorphosing offspring were consumed by Brewer's blackbirds (Bradford 1991).
Introduction of non-native fishes are a major threat to this species. Prior to stocking, fish were generally historically absent from the mid to high elevations in the Sierra Nevada (Hayes and Jennings 1986, Bradford et al. 1993, Knapp 1996). Both distribution and abundance of mountain yellow-legged frog larvae are significantly reduced when trout are introduced to an area (Knapp et al. 2001). When fish are removed from an area, frog populations immediately begin to recover regardless of other habitat conditions (Knapp et al. 2001; Knapp et al. 2007). Additionally, when fish are removed, the larvae numbers mirror larvae numbers in lakes where fish were never introduced (Knapp et al. 2001).

The long larval stage for the Sierra Nevada yellow-legged frog makes it extremely vulnerable to predation by fish, where it must overwinter under ice generally two to three times before metamorphosis (Bradford 1989, Vredenburg et al. 2005). Finlay and Vredenburg (2007) found that densities of larvae and frogs were significantly higher in fishless lakes than those with trout. Fish also greatly reduce the availability of prey to adult frogs, which only forage on aquatic invertebrates when they are at the water surface or near the shoreline (Finlay and Vredenburg 2007).

Disease is a major source of concern for mountain yellow-legged frogs. Two diseases are particularly hard on this species. The first is known as "red-leg" disease and is caused by the bacterium *Aeromonas hydrophila*. Animals with this disease are emaciated, sluggish, poorly coordinated and the ventral surfaces of limbs are abnormally red due to hemorrhage and enlarged capillaries. "Red-leg" disease is attributed to the die-off of approximately 800 adult frogs at a single location over the timespan of a single season (Bradford 1991). It should be noted that although "red-leg" disease is attributed to that particular die-off, the diagnosis was made before amphibian chytridiomycosis was well known and the die off may have been the result of a combination of both diseases or the result of only one of the two diseases. This second disease, amphibian chytridiomycosis, is caused by the fungus *Batrachochydrium dendrobatidis* (Bd). Chytridiomycosis is an emerging infection disease which has caused numerous declines and possible extinctions of amphibians globally. Mountain yellow-legged frogs are well documented as being sensitive to this disease. Animals are able to acquire Bd zoospores by simply being in an infected lake, frog-frog contact is not required (Rachowicz and Briggs 2007). Although Bd is considered a primary cause for many of the disappearances of mountain yellow-legged frogs, some populations are able to coexist with the fungus. These populations have a significantly larger proportion of anti-Bd bacterial species than populations that went extinct shortly after the appearance of Bd in the area. This is indicative of herd immunity where populations with a high proportion of individuals protected by bacteria limits the survival of the disease and thus prevents epidemic outbreaks in that population (Lam et al. 2010; Woodhams et al. 2007). At least 83% of all known sites currently have Bd present (Knapp et al. 2011).

Sierra Nevada yellow-legged frogs are known to have been present within a number of locations in the Tahoe National Forest, but now exist in only a few populations in ponds and streams and generally in small numbers (USFWS 2003, the Tahoe National Forest GIS database). Jennings and Hayes (1994) indicate that the species was extinct by 1992 in a number of locations based on re-surveys of historic locations.

The Tahoe National Forest initiated herpetological surveys in 1996 in cooperation with the California Academy of Sciences, which included areas likely to support mountain yellow-legged frogs. These surveys continued through 1999, and included a systematic search of historical museum records for the four counties encompassing the Tahoe National Forest (Vindum et al. 1997, Vindum and Koo 1999a, Vindum and Koo 1999b). The review of historical herpetological specimens found that mountain yellow-legged frogs were historically collected from 33 localities in the Tahoe National Forest (Vindum et al. 1997). During ensuing surveys from 1997-1999, Sierra Nevada yellow-legged frogs were found in two additional localities (Vindum et al. 1997, Vindum and Koo 1999a, Vindum and Koo 1999b). Mountain yellow-legged frog surveys were also conducted in cooperation with the USGS Biological Division, Pt. Reyes, from 1997 through 2000, and continue periodically (data on file with the Tahoe National Forest). Since 1997, mountain yellow-legged frog sightings have been routinely recorded, either incidentally during stream and other biological surveys or during amphibian-focused surveys.

The Tahoe National Forest GIS database shows that since 1993 there have been mountain yellow-legged frogs documented in 4 general localities on Truckee Ranger District, 6 general localities on Sierraville Ranger District, and 10 general localities on Yuba River Ranger District.

The Forest-wide Standards and Guidelines associated with Riparian Conservation Areas (Nos. 91-94) and those associated with Riparian Conservation Objectives (Nos. 95-124) in the SNFPA ROD (USDA Forest Service 2004) are intended to maintain the function and integrity of riparian habitats upon which the mountain yellow-legged frog relies.

Two sections of trail re-routes cross perennial drainages above 4500 feet in elevation. Suitable pool breeding habitat is not present at either of these crossings, or for 1 km above or below it. This species has not been identified to occur within any drainages that cross through this project area—Lavezzola, Rattlesnake, or the Downie River—or any of their tributaries. There are no high elevation lakes or ponds that are present within severak km. of this crossing. Additionally, this species is primarily found at elevations above 6,000 feet in elevation. All other project activities either occur below the elevational range of this species, or outside of suitable habitat. There is no critical habitat in the project area.

B. Sierra Nevada Yellow-Legged Frog: Effects of the Proposed Action and Alternatives including Project Design Standards

The following Management Requirement will protect this species:

1. Conduct pre- implementation surveys in Rattlesnake Creek prior to implementing any ground disturbing activities within 82 feet of the drainage. If frogs are seen, stop work immediately, and notify an aquatic biologist to determine appropriate actions.

2. To avoid impacts to aquatic species, construct bridges across the perennial crossing at Rattlesnake Creek and at the bottom of the re-route across Lavezzola Creek.

Direct and Indirect Effects

Individual frogs could be disturbed or injured during trail construction within habitat, or from trail use, if a wet stream crossing occurs. Howevr, the project Management Requirements would avoid these effects from occurring during project implementation, and from longer term use of the trail itself.

Cumulative Effects

Because there are no direct or indirect effects, there are no cumulative effects that would occur.

C. Mountain Yellow-Legged Frog: Conclusion and Determination

It is my determination that implementation of this project will not affect the Sierra Nevada yellow-legged frog.

GREAT BASIN RAMS-HORN SNAIL

Status: USFS R5 Sensitive

A. Great Basin Rams-Horn Snail: Existing Environment

The Great Basin rams-horn snail is listed as Sensitive on the Region 5 Forester's Sensitive Species List (USDA Forest Service 1998). The Tahoe National Forest LRMP, as amended, does not provide specific management guidelines for this species. SNFPA standards and guidelines for Riparian Conservation Areas, listed under California red-legged frog, provide for the needs of this species.

The Great Basin ramshorn snail occurs in a highly restrictive distribution but is locally abundant. Historically, the Great Basin ramshorn snail occurred within the lakes and larger, slow streams in and around the northern Great Basin. In California the snail was known to occur in six local drainages in which the species probably survives in four of these drainages.

The Great Basin rams-horn snail occurs in larger lakes and slow rivers including larger spring sources and spring-fed creeks. These snails characteristically burrow in soft mud and may be invisible even when abundant (Taylor 1981). The Great Basin ramshorn snail can occur with *Pisidium ultramontanum, Lanx klamathensis,* or several other endemic molluscs (Frest and Johannes 1993). It also occurs with *Juga acutifilosa* and *Fluminicola seminalis.* Habitat requirements include cold highly oxygenated water, muddy substrate, and slow stream flow. Springs are preferred, but the snail will use river margins. Soft sediments are preferred. Threats to snails have been attributed to water diversions and water pollution. Mitigations for fish species, such as adding spawning gravels, may harm this species by smothering soft mud habitats.

Threats to this species include but are not limited to habitat alteration, changes in water flow regime, changes in water quality and loss of hosts for development.

Historically, the Great Basin rams-horn snail has been observed in the Truckee River directly downstream of Lake Tahoe, on the Lake Tahoe Basin Management Unit. Currently, this snail has not been sighted or surveyed for in the Tahoe National Forest. Suitable habitat occurs within slow segments of the Truckee and Little Truckee Rivers and their tributaries.

The geographic range of this species is outside of the Yuba River Ranger District.

B. Great Basin Rams-Horn Snail: Effects of the Proposed Action and Alternatives including Project Design Standards

Because this project is outside of the geographic range of this species, there are no direct, indirect, or cumulative effects that would occur.

C. Great Basin Rams-Horn Snail: Conclusion and Determination

It is my determination that this project will not affect the Great Basin rams-horn snail.

LAHONTAN LAKE TUI CHUB

Status: USFS R5 Sensitive

A. Lahontan Lake Tui Chub: Existing Environment

EAST OF THE SIERRA NEVADA MOUNTAIN CREST ONLY

The Lahontan Lake tui chub is listed as Sensitive on the Region 5 Forester's Sensitive Species List (USDA Forest Service 1998). The Tahoe National Forest LRMP, as amended, does not provide specific management guidelines for this species. SNFPA standards and guidelines for Riparian Conservation Areas, listed under California red-legged frog, provide for the needs of the Lahontan Lake tui chub.

The Lahontan Lake tui chub are a cyprinid subspecies found in Lake Tahoe and Pyramid Lake (Nevada) which are connected to each other by the Truckee River and in nearby Walker Lake (Nevada). Populations of plankton-feeding chub occurring in Stampede, Boca and Prosser reservoirs may also be Lahontan Lake tui chub due to morphological similarities (Marrin and Erman 1982, Moyle et al. 1995).

The Lake Tahoe population is the only confirmed population in the Sierra Nevada, with a probable population in Stampede, Boca and Prosser Reservoirs in the Tahoe National Forest. Little study has occurred on the Lake Tahoe population since Miller (1951). Zooplankton levels have changed over this period. *Daphnia*, an important prey of adult chubs, have been nearly eliminated (Richards et al. 1975) by the introduced Kokanee salmon (*Oncorhynchus nerka*) and opossum shrimp (*Mysis relicta*), both of which feed on zooplankton. Marshland degradation along the lake may be taking away vital spawning and nursery areas.

Based on occurrence within such widely diverse habitats as Lake Tahoe and Pyramid Lake, it is believed this species can tolerate a wide range of physicochemical water conditions. Lahontan Lake tui chub are known as a mid-water feeder. In Lake Tahoe, larger fish (>16 cm TL) occur in deeper water (>50m) during the day, moving into shallower water areas at night (Miller 1951). Young fish generally occur in shallow water. It has also been noted that a seasonal migration occurs within the water column. Deeper water in often utilized during winter months and summer months show use of upper portions (Snyder 1917, Miller 1951). Algal beds in shallow inshore areas seem necessary for spawning, egg hatching, and larval survival.

Lahontan Lake tui chub are schooling fish reaching lengths of 35 to 41 cm FL, which inhabit large, deep lakes (Moyle 1976). Lahontan Lake tui chub feed primarily on zooplankton, especially cladocerans and copepods, but also eat benthic insects when available (Miller 1951, Marrin and Erman 1982). Tui chub are predated upon mostly by large trout, and rarely by birds and snakes (Miller 1951).

In Lake Tahoe, nocturnal spawning occurs during May and June, possibly extending into July (Miller 1951). Tui chub may be serial spawners, reproducing several times during the spawning season (Moyle 1976). Reproductive adults spawn near-shore over beds of aquatic vegetation, to which the eggs adhere (Snyder 1917). Young remain near-shore until winter when body size is 1-2 cm; then migrate into deeper water. Linear growth of tui chubs occurs about 4 years; then mass is accumulated rapidly. The largest documented length in Lake Tahoe is 13.7 cm SL, but longer chub (21 cm) have been found in Walker Lake, Nevada (Miller 1951).

Threats to the Lahontan Lake tui chub include but are not limited to water quality, specifically alkalinity due to diversions of inflowing water, change in prey base due to introduced species, and reservoir and wetland management.

Surveys for this species have not been conducted in the Tahoe National Forest. Populations of plankton-feeding chub occur in Stampede, Boca and Prosser Reservoirs; these may be Lahontan Lake tui chub due to morphological similarities (Marrin and Erman 1982, Moyle et al. 1995).

Potential risk factors include but are not limited to water quality, specifically alkalinity due to diversions of inflowing water, change in prey base due to introduced species, and reservoir and wetland management.

This project does not occur within the geographic range of this species.

B. Lahontan Lake Tui Chub: Effects of the Proposed Action and Alternatives including Project Design Standards

Because this project is outside of the range of this species, there are no direct, indirect, or cumulative effects that would occur.

C. Lahontan Lake Tui Chub: Conclusion and Determination

It is my determination that this project will not affect the Lahontan Lake tui chub.

HARDHEAD

Status: USFS R5 Sensitive

A. Hardhead: Existing Environment

The hardhead is listed as Sensitive on the Region 5 Forester's Sensitive Species List (USDA Forest Service 1998). The Tahoe National Forest LRMP, as amended, does not provide specific management guidelines for this species. SNFPA standards and guidelines for Riparian Conservation Areas, listed under California red-legged frog, provide for the needs of this species.

Historically, hardhead have been regarded as a widespread and locally abundant species (Ayres 1854, Jordan and Evermann 1896, Evermann 1905, Rutter 1908, Murphy 1947, Soule 1951, Reeves 1964). Hardhead are still widespread in the foothill streams, but their specialized habitat requirements, combined with widespread alteration of downstream habitats have resulted in isolation and localization of populations. These conditions increase the chance for localized extinctions. Hence, hardhead are less abundant than they once were, especially in the southern half of their range.

Hardhead are widely distributed in low to mid-elevation streams in the main Sacramento-San Joaquin drainage as well as the Russian River drainage. Hardhead are typically found in undisturbed areas of larger middle- and low-elevation streams (up to 4,390 feet) (Moyle and Nichols 1973, Moyle 1976). Their range extends from the Kern River, Kern County, in the south to the Pitt River, Modoc County, in the north. Populations are scattered in the tributary streams of the San Joaquin drainage, but have not been found in the valley reaches of the San Joaquin River (Moyle and Nichols 1973, Saiki 1984, Brown and Moyle 1987). In the Sacramento River drainage, hardhead are present in most of the larger tributary streams as well as the Sacramento

River. They are widely distributed in the Pitt River drainage (Cooper 1983, Moyle and Daniels 1982), including the main Pitt River and its series of Hydroelectric reservoirs.

Hardhead were first described by Baird and Girard (Girard 1854) as *Gila conocephala.* Ayres (1854) redescribed it as *Mylopharodon robustus*. *G. conocephala* was later classified as *M. conocephalus* and considered to be closely related to *M. robustus*. Electrophoresis studies by Avise and Ayala (1976) indicate the hardhead to be closely related to the Sacramento squawfish, but different enough to be considered a separate species.

Most streams occupied by hardhead have summer temperatures in excess of 20°C, selecting an optimal range between 24-28°C (Knight 1985). Hardhead are relatively intolerant of low oxygen levels, especially at higher temperatures, a factor which may limit their distribution to well oxygenated streams and the surface water of reservoirs (Cech et al. 1990). They prefer clear, deep (> 1m) pools with sand-gravel-boulder substrates and slow water velocities (<25cm sec^{-1}) (Moyle and Nichols 1973, Knight 1985, Moyle and Baltz 1985). In streams, adult hardhead tend to remain in the lower half of the water column, rarely moving into the upper levels (Knight 1985), while juveniles concentrate in shallow water close to the stream edges (Moyle and Baltz 1985). Hardhead are always found in association with Sacramento squawfish and usually with Sacramento suckers. They tend to be absent from streams introduced with exotics, especially centrarchids (Moyle and Nichols 1973, Moyle and Daniels 1982), or streams that have been severely altered by human activity (Baltz and Moyle 1993).

Primarily bottom feeders, hardheads forage for benthic invertebrates and aquatic plant material in quiet water. They will occasionally feed on plankton and surface insects. Smaller fish (<20 cm SL) feed primarily on mayfly larvae, caddisfly larvae, and small snails, whereas larger fish feed mainly on aquatic plants (especially filamentous algae), as well as crayfish and other large invertebrates (Reeves 1964). The ontogenetic changes in teeth structure seem to coincide with dietary changes. Reeves (1964) found no remains of fish in the stomachs of large hardhead.

Hardhead mature after their second year and most likely spawn in the spring (Reeves 1964), judging by the by the upstream migrations of adults into smaller tributary streams during this time of year (Wales 1946, Murphy 1947, Bell and Kimsey 1955, Rowley 1955). Estimates based on juvenile recruitment suggest that hardhead spawn by May-June in Central Valley streams and that the spawning season may extend into August in the foothill streams of the Sacramento-San Joaquin drainage (Wang 1986).

Hardhead reach 7-8 cm by their first year, but growth slows in subsequent years. In the American River, hardhead reach 30 cm SL in 4 years, whereas in Pitt and Feather Rivers, it takes six years to reach that length (Moyle et al. 1983, PG&E 1985).

Potential risk factors include but are not limited to widespread alteration of down stream habitats, population isolation that increases possibility of local extinction, habitat loss from hydroelectric power developments, and predation by exotic species.

Hardhead are known to occur within the main stem of the North Yuba River. They do not occur within headwater tributary drainages of the Yuba, which is where this project occurs. Therefore, hardhead nor their habitat occur within this project area.

B. Hardhead: Effects of the Proposed Action and Alternatives including Project Design Standards

Due to the lack of suitable habitat, there are no direct, indirect, or cumulative effects that would occur from this project.

C. Hardhead: Conclusion and Determination

It is my determination that this project will not affect the hardhead.

CALIFORNIA FLOATER

Status: USFS R5 Sensitive

A. California Floater: Existing Environment

The California floater (*Anodonta californiensis*) is listed as a Sensitive Species in Region 5 and designated as a “species of special concern” by the State of California. Standards and guidelines in the Sierra Nevada Forest Plan Amendment for Riparian Conservation Areas, listed under California red-legged frog, provides for the needs of the California floater. The California floater is a freshwater mussel found in lakes and slow rivers (Taylor 1981).

The type locality of this species is the “Rio Colorado’, a former distirbutary of the river, New River, Imperial County, CA (Taylor 1981). Historically this species was distributed on the lower Willamette and lower Columbia rivers in Oregon and Washington. It occurred in larger slow streams of northern California south to northern San Joaquin Valley. It’s former range in California includes Siskiyou, Shasta, Lassen, Modoc and Tehama Cos. (Frest and Johannes 1995).

It’s current distribution indicates this species has probably been eliminated from much of its former range (Taylor 1981). It is apparently extinct in the Upper Sacramento River and appears to be extinct in Utah with a limited distribution in Arizona. The current known distribution in California are the Lassen, Modoc, and Shasta-Trinity National Forests. This species still survives in Fall and Pit Rivers, Shasta Co. *Anodonta californiensis* has been reported to occur adjacent to the Tahoe National Forest, but no occurrences have been documented on National Forest system lands within the boundary of the Tahoe National Forest. Donner Lake is reported as the locality of an unconfirmed historic sighting in a mollusk database created by Dr. Jayne Brim-Box and Jeff Kershner. The species has been reported to occur at the following sites in Nevada: 1)Truckee River, 2) Humboldt River, Humboldt Basin, Elko, Co in 1979, 3) Thousand Springs Valley northeast of Wells, Elko Co., Lake Bonneville Basin in 1989 (Nevada Natural Heritage Database).

Anodonta californiensis occurs in lakes and slow rivers (Taylor 1981), generally, on soft substrates (mud-sand), in fairly large streams and lakes, in relatively slow currents (Frest and Johannes 1995). Howard and Cuffey (2003) found that *A. californiensis* was almost exclusively found in pools with no riffles and very few in runs in the south Fork of the Eel River in Oregon.

This species range does not include the Yuba River Ranger District.

B. California Floater: Effects of the Proposed Action and Alternatives including Project Design Standards

Because this species range does not include the Yuba River Ranger District, there are no direct, indirect, or cumulative effects that would occur.

C. California Floater: Conclusion and Determination

It is my determination that this project will not affect the California floater.

BLACK JUGA

Status: USFS R5 Sensitive

A. Black Juga: Existing Environment

NatureServe (http://www.natureserve.org/explorer/, updated January, 2000) reports the conservation status of Juga nigrina (Lea 1856) as G3, vulnerable. Taylor (1981) listed one synonym, Melania californica (Clessin 1882) for *Juga nigrina*. He described the species as commonly occurring in tributaries of the Sacramento River and interior drainages of northeastern California, locally in the upper Klamath River, the uppermost Eel River drainage, the Napa River and coastal streams of Mendocino County (Big and Noyo rivers) and south into the Russian River drainage of Sonoma County with the southern-most population in Salmon Creek of Sonoma County apparently extirpated. Historically, black juga Juga nigrina was described as occurring in headwater streams and river tributaries from northwestern California to southwestern Oregon (Henderson 1929, Burch 1989). However, the genetic sequence attributed to J. nigrina from the Umpqua basin in Oregon by Holznagel and Lydeard (2000), was found by Campbell et al. (2010) to be more closely related to specimens of J. silicula from the Willamette River. The family Pleuroceridae, to which this species belongs, is currently the most diverse in North America with about 1,000 nominal and 200 valid species (Strong et al. 2008). Two closely related species, Juga acutifilosa and occata, are Pacific Southwest Region Sensitive Species.

Recent analyses (Campbell et al. 2010) based on anatomy and genetics established that this species, as recognized in museum collections and literature, is composite. According to Frest & Johannes (1995), Juga nigrina occurs in the upper Sacramento, McCloud and Pit river systems. They reported collecting this species from 31 of 231 sites surveyed in the upper Sacramento and Pit river systems (see Figure 1) and concluded that the species had been extirpated from a “fair number” of historic sites in tributaries to the upper Sacramento River.

Therefore, black juga, as presently understood taxonomically, is restricted to the upper Sacramento system in California. The type-locality populations in Clear Creek, Shasta County, tributary to the Sacramento River, have been decimated by gold mining activities (Frest and Johannes 1995), but the species still persists in Clear Creek above the town of French Gulch, the epicenter for the mining operations (Johannes 2010). The species only occurs in California.

Frest & Johannes (1995) report that the species has been extirpated from most of its type locality of Clear Creek, tributary to the upper Sacramento River, due to gold mining. The authors further concluded that this species has been extirpated from several sites based on the apparent absence of this species at many historic sites in the upper Sacramento River system (Frest & Johannes 1995, See Table 10 for a list of museum records.)

Threats to spring and stream habitats occupied by Juga nigrina include:

- Excessive sedimentation from a variety of activities such as mining, logging, road and railroad grade construction, and grazing may smother substrates and stress or kill individuals, and impair egg-laying or survivorship of eggs or young.
- Livestock trampling and grazing of small streams, springs and spring runs resulting in reduced dissolved oxygen levels, or elevated fine sediments and water temperature.
- Water diversions resulting in reduced spring or stream flow, elevated water temperatures, fine sediment accumulations, lower dissolved oxygen and thus less suitable habitat.
- Dam construction, which inundates cold springs, slows current velocities, lowers the availability of oxygen and allows fine sediments to accumulate.
- Excessive sedimentation from a variety of activities such as logging, mining, road and railroad grade construction, and grazing may smother substrates and stress or kill individuals, and impair egg-laying or survivorship of eggs or young.

There are no historical records of this species occurring within the Yuba River. Although this species historic range does not include the Yuba River drainage, it is possible that it is in tributary headwaters in the project area.

B. Black Juga: Effects of the Proposed Action and Alternatives including Project Design Standards

None of the proposals occur within drainages, or would affect any habitat for this species. Therefore, there are no direct, indirect or cumulative effects that would occur.

C. Black Juga: Conclusion and Determination

It is my determination that this project will not affect the black juga snail.

VII. MANAGEMENT REQUIREMENTS

The following Management Requirements are included in the proposed action, and must be implemented in full, or it may invalidate the determination statements in this Biological Evaluation, and it may necessitate additional consultation with the U. S. Fish and Wildlife Service.

1. To avoid disturbances to spotted owls during the breeding season, Limit the Operating Period from March 1 through August 15 for trail construction and decommissioning along the Lavezzola Trail re-route, unless surveys are conducted to protocol, and it is determined that this is no longer needed.

2. To avoid disturbances to northern goshawks during the breeding season, Limit the Operating Period from February 15 through September 15 for trail construction and decommissioning along the Butcher Ranch Trail re-route, unless surveys conducted to protocol determine that this is no longer necessary.

3. To avoid impacts to aquatic species, construct a bridge across the perennial crossing at Rattlesnake Creek and at the bottom of the re-route across Lavezzola Creek.

4. Where practicable, locate trail to avoid the need to fall large trees and snags, or those displaying wildlife use (cavities, nests). Fall and leave hazardous snags to recruit dead wood.

5. If new Threatened, Endangered, or Forest Service Sensitive (TES) species are listed or discovered, or nesting TES are found within 0.25 mile of activities, a limited operating period will be implemented as recommended by a qualified biologist.

6. Implement project-specific Best Management Practices that include measures to minimize sedimentation into creeks and maintain water quality.

VIII. MANAGEMENT RECOMMENDATIONS

There are no management recommendations specific for fish and wildlife for this project.

IX. LITERATURE CITED AND REFERENCES

Literature Cited (USDA Forest Service, 2004) and References are first listed by General Resource Documents, followed by species-specific or familial group Literature Cited and References.

General Resource Documents

Endangered Species Act. 1973 as amended. 50 CFR Chapter IV part 402. 12pp.

National Environmental Policy Act. 1986. 40 CFR Parts 1500-1508. 45pp.

Sierra Nevada Ecosystem Project, Final Report to Congress. 1996. Volumes 1-111 and addendum.

Tahoe National Forest Land and Resources Management Plan. 1990.

Tahoe National Forest maps for Endangered, Threatened and Sensitive species sightings and their habitats.

USDA Forest Service. 1984. Land Management Planning Direction. Pacific Southwest Region. San Francisco.

USDA Forest Service. 2013. Sensitive Species List 2013. Pacific Southwest Region. San Francisco.

USDA Forest Service. 1998. Sierra Nevada Science Review--report of the Science Review Team charged to synthesize new information of range wide urgency to the national forests of the Sierra Nevada. Pacific Southwest Research Station. July 25, 1998. 115pp.

USDA Forest Service. 1998. Region 5 letter, July 31. Use of information in the Sierra Nevada Science Review in planning and decision making.

USDA Forest Service. 2000. Washington Office letter, March 13. Direction on Forest Service timber sale environmental analysis requirements.

USDA Forest Service. 2001. Sierra Nevada Forest Plan Amendment: Final Environmental Impact Statement, vols 1-6, including appendices, and Record of Decision. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, California.

USDA Forest Service. 2003. Sierra Nevada Forest Plan Amendment: Management Review and Recommendations. R5-MB-012. March 2003. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, California.

USDA Forest Service. 2004. Sierra Nevada Forest Plan Amendment: Final Supplemental Environmental Impact Statement, vols 1-2 and Record of Decision. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, California.

U. S. Fish and Wildlife Service (USFWS). Letter to Tahoe National Forest with species list. Available at: Tahoe National Forest Supervisor's Office, Nevada City, CA.

U. S. Fish and Wildlife Service (USFWS). 2001. (Letter from Wayne S. White to Bradley Powell and Jack Blackwell, January 11; Biological Opinion on Formal Endangered Species Consultation and Conference on the Biological Assessment for the Sierra Nevada Forest Plan Amendment Final Environmental Impact Statement.) Copy located at: Tahoe National Forest, Supervisor's Office, Nevada City, CA.

Western Bumble Bee

Alaux, C., J. Brunet, C. Dussaubat, F. Mondet, S. Tchamitchan, M. Cousin, J. Brillard, A. Baldy, L.P. Belzunces and Y. Le Conte. 2010. Interactions between Nosema microspores and a neonicotinoid weaken honeybees (*Apis mellifera*). Environmental Microbiology 12: 774–78.

Cameron, S.A., J.D. Lozier, J.P. Strange, J.B. Koch, N. Cordes, L.F. Solter and T.L. Griswold. 2011. Patterns of widespread decline in North American bumble bees. Proceedings of the National Academy of Sciences 108:662-667. See http://www.pnas.org/content/108/2/662.full.pdf+html .

Carvell, C., W.R. Meek, R.F. Pywell, D. Goulson, M. Nowakowski. 2007. Comparing the efficacy of agri-environment schemes to enhance bumble bee abundance and diversity on arable field margins. Journal of Applied Ecology, 44:29-40.

Dupont, Y.L, C. Damgaard, V. Simonsen. 2011. Quantitative Historical Change in Bumblebee (*Bombus* spp.) Assemblages of Red Clover Fields. PuPLoS One Volumbe 6: Issue 9. Available at http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0025172

Evans, E., R. Thorp, S. Jepsen and S.H. Black. 2008. Status Review of Three Formerly Common Species of Bumble Bee in the Subgenus Bombus: *Bombus affinis* (the rusty patched bumble bee), *B. terricola* (the yellowbanded bumble bee), and *B. occidentalis* (the western bumble bee). The Xerces society, Portland, OR. Available at http://www.xerces.org/wp-content/uploads/2009/03/xerces_2008_bombus_status_review.pdf

Goulson, D., G.C. Lye and B. Darvill. 2008. Decline and Conservation of Bumble Bees. Annual Review of Entomology 53:191–208

Hatfield, R. 2012. Records of western and Franklin's bumble bees in the western United States. Database records provided by the Xerces Society, Portland, OR on 2/29/12.

Heinrich, B. 1979. *Bumblebee Economics*. Harvard University Press, Cambridge, MA. 245 pp.

Henry,M., M. Beguin, F. Requier, O. Rollin, J. Odoux, P. Aupinel, J. Aptel, S. Tchamitchian and A. Decourtye. 2012. A Common Pesticide Decreases Foraging Success and Survival in Honey Bees. SciencExpress available at http://www.sciencemag.org/content/early/2012/03/28/science.1215039.full.pdf

Hopwood, J., M. Vaughan, M. Shepherd, D. Biddinger, E. Mader, S. Hoffman Black and C. Mazzacano. 2012. Are Neonicotinoids Killing Bees? A Review of Research into the Effects of Neonicotinoid Insecticides on Bees, with Recommendations for Action. Xerces Society, Portland, OR. Available at http://www.xerces.org/wp-content/uploads/2012/03/Are-Neonicotinoids-Killing-Bees_Xerces-Society1.pdf .

Koch, J., J. Strange and P. Williams. 2012. Bumble Bees of the Western United States. U.S. Forest Service and the Pollinator Partnership, Washington, D.C. 144 pp.

Kreyer, D., A. Oed, K. Walther-Hellwig and R. Frankl. 2004. Are forests potential landscape barriers for foraging bumblebees? Landscape scale experiments with *Bombus terrestris* agg. and *Bombus pascuorum* (Hymenoptera, Apidae). Biological Conservation 116 :111–118.

Loffland, H. L., Polasik, J. S., Tingley, M. W., Elsey, E. A., Loffland, C., Lebuhn, G. and Siegel, R. B. (2017), Bumble bee use of post-fire chaparral in the central Sierra Nevada. Jour. Wild. Mgmt., 81: 1084–1097. doi:10.1002/jwmg.21280

Mader, E., M. Shepherd, M. Vaughan, S. Black and G. LeBuhn. 2011. Attracting Native Pollinators: Protecting North America's Bees and Butterflies. The Xerces Society Guide to Conserving North American Bees and Butterflies and Their Habitat. Storey Publishing, North Adams, MA. 371 pp.

McFrederick QS, Le Buhn G. 2006. Are urban parks refuges for bumble bees? Biol. Conserv. 129:372–382.

Mikkola, K. 1984. Migration of wasp and bumblebee queens across the Gulf of Finland (Hymenoptera: Vespidae and Apidae). Notulae Entomologicae 64: 125-128.

Osborne, J.L., A.P. Martin, N.L. Carreck, J.L. Swain, M.E. Knight, D. Goulson, R.J. Hale and R.A. Sanderson. 2008. Bumblebee flight distances in relation to the forage landscape. Journal of Animal Ecology 77:406–415.

Rao,S., W.P. Stephen, C. Kimoto and S.J. DeBano. 2011. The Status of the ‘Red-Listed’ *Bombus occidentalis* (Hymenoptera: Apiformes) in Northeastern Oregon. Northwest Science 85: 64-67.

Saab, V. A. and H. D. W. Powell. 2005. Fire and avian ecology in North America: Process influencing pattern. Studies in Avian Biology 30:1-13.

Schweitzer, D.F., N.A. Capuano, B.E. Young, and S.R. Colla. 2012. Conservation and management of North American bumble bees. NatureServe, Arlington, Virginia, and USDA Forest Service, Washington, D.C.

Thorp, R. W., and M. D. Shepherd. 2005. Profile: Subgenus *Bombus*. In Shepherd, M. D., D. M. Vaughan, and S. H. Black (Eds). Red List of Pollinator Insects of North America. The Xerces

Society for Invertebrate Conservation, Portland, OR.

Tommasi, D., A. Miro, H. A. Higo and M. L. Winston. 2004. Bee diversity and abundance in an urban setting. The Canadian Entomologist 136: 851–869.

Williams, P.H. and J.L. Osborne. 2009. Bumblebee vulnerability and conservation world-wide. Apidologie 40:367–387.

Bald Eagle

Alt, K. L. 1980. Ecology of the nesting bald eagle and osprey in the Grand Teton-Yellowstone National Parks Complex. M.S. Thesis, Montana State Univ., Bozeman. 95 pp.

Andrew J. M. and J. A. Mosher. 1982. Bald eagle nest site selection and nesting habitat in Maryland. Journal of Wildlife Management 46:383-390.

Anthony, R. G. and F. B. Isaacs. 1989. Characteristics of bald eagle nest sites in Oregon. Journal of Wildlife Management 53:148-159.

Anthony, R. G., R. L. Knight, G. T. Allen, B. R. McClelland and J. I. Hodges. 1982. Habitat use by nesting and roosting bald eagles in the Pacific Northwest. Trans. A. Am. Wildl. Natural Res Conf. 47:332-342.

Bald and Golden Eagle Protection Act. 1940. P.L. 76-567, Ch. 278, 54 Stat. 250, as amended; 16 U.S.C. 668 (note), 668, 668a-d). June 8.

Buehler, D. A., T. J. Mersmann, J. D. Fraser, J. K. D. Seegar. 1991. Effects of human activity on bald eagle distribution on the northern Chesapeake Bay. Journal of Wildlife Management 55:282-290.

Craig, D. 1991. A habitat management program for the bald eagle on the Tahoe National Forest—draft; unpublished report. Available at the Tahoe National Forest Supervisor's Office, Coyote Street, Nevada City, CA. 10p.

Dietrich, P. J. 1981. Historic range of breeding bald eagles in California. Unpublished manuscript. Redding, CA. 17 pp.

Detrich, P. J. 1982. Results of California winter bald eagle survey--1982. U. S. Dept. Int. Fish Wildl. Serv., Sacramento, CA. 16 pp.

Detrich, P. 1990. The California bald eagle management plan—draft; unpublished report. Available at the Tahoe National Forest Supervisor's Office, Coyote Street, Nevada City, CA. 71p.

Fraser, J. D. 1985. The impact of human activities on bald eagle populations—a review. Pages 68-84 *in* J.M. Gerrard and T.M. Ingram, eds. The bald eagle in Canada. Headingley, Manitoba: White Horse Plains Pub.

Fraser, J. D., L. D. Frenzel, J. E. Mathisen. 1985. The impact of human activities on breeding bald eagles in north-central Minnesota. Journal of Wildlife Management 59:585-592.

Grubb, T. G. 1976. A survey and analysis of bald eagle nesting in Western Washington. M.S. Thesis, Univ. Washington, Seattle. 87 pp.

Grubb, T. G. 1995a. Constructing bald eagle nests with natural materials. USDA Forest Service research note RM-RN-535, 8p. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Grubb, T. G. 1995b. Food habits of bald eagles breeding in the Arizona Desert. Wilson Bulleton 107:258-274.

Grubb, T. G., W. W. Bowerman, J. P. Giesy and G. A. Dawson. 1992. Responses of breeding bald eagles, *Haliaeetus leucocephalus*, to human activities in northcentral Michigan. Canadian Field-Naturalist 106(4):443-453.

Grubb, T. G. and C. E. Kennedy. 1982. Bald eagle winter habitat on southwestern Nationl Forests. USDA Forest Service Research paper RM-237, 13p. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Grubb, T. G. and R. M. King. 1991. Assessing human disturbance of breeding bald eagles with classification tree models. Journal of Wildlife Management 55:500-511.

Jurek, R. M. 1988. Five-year status report—bald eagle. State of California, The Resources Agency, Department of Fish and Game; unpublished report. 15p.

Keister, G. P. and R. G. Anthony. 1983. Characteristics of bald eagle communal roosts in the Klamath Basin, Oregon and California. Journal of Wildlife Management 47:1072-1079.

Knight, R. L. and S. K. Skagen. 1987. Effects of recreational disturbance on birds of prey: a review. Southwest Raptor Management Symposium and Workshop.

Lehman, R. N. 1979. A survey of selected habitat features of 95 bald eagle nest in California. Calif. Dept. Fish and Game. Wildl. Manage. Branch Admin. Rep. 79-1, Sacramento. 23 pp.

Lehman, R. N., D. E. Craigie, P. L. Colins and R. S. Griffen. 1980. An analysis of habitat requirements and site selection criteria for nesting bald eagle in California. Report by Wilderness Research Institute, Arcata, CA. for U.S. Forest Service, Region 5, San Francisco, CA. 106 pp.

Manci, K. M. D. N. Gladwin, R. Villella, and M. G. Cavendish. 1988. Effects of aircraft noise and sonic booms on domestic animals and wildlife: a literature synthesis. U.S. Fish and Wildlife Service, National Ecology Research Center, Fort Collins, CO. 88p.

Montopoli, G. J. and D. A. Anderson. 1991. A logistic model for the cumulative effects of human intervention on bald eagle habitat. J. Wildl. Manage. 55:290-293.

Stalmaster, M. V. 1976. Winter ecology and effects of human activity on bald eagles in the Nooksack River Valley, Washington. M.S. Thesis. West. Washington State Coll. Bellingham, WA. 100 pp.

Stalmaster, M. V. and J. R. Newman. 1978. Behavioral responses of wintering bald eagles to human activity. J. Wildl. manage. 52:506-513.

USDA Forest Service. 1977. Bald eagle habitat management guidelines, unpublished report, U. S. Forest Service, California Region. 60p.

U. S. Fish and Wildlife Service (USFWS). 1967. Endangered Species. Federal Register 32(48):4001.

U. S. Fish and Wildlife Service (USFWS). 1978. Determination of Certain Bald Eagle Populations as Endangered or Threatened. Federal Register 43(31):6230-6233.

U. S. Fish and Wildlife Service (USFWS). 1986. Pacific Bald Eagle Recovery Plan. U. S. Fish and Wildlife Service, Portland, Oregon. 160 pp.

U. S. Fish and Wildlife Service (USFWS). 1995. Final Rule to Reclassify the Bald Eagle From Endangered to Threatened in All of the Lower 48 States. Federal Register 60(133):36000-36010.

U. S. Fish and Wildlife Service (USFWS). 2007a. National Bald Eagle Management Guidelines. U. S. Fish and Wildlife Service. 23pp.

U. S. Fish and Wildlife Service (USFWS). 2007b. Protection of Eagles; Definition of "Disturb". Federal Register 72(107):31132-31140.

U. S. Fish and Wildlife Service (USFWS). 2007c. Authorizations Under the Bald and Golden Eagle Protection Act for Take of Eagles (proposed rule). Federal Register 72(107):31141-31155.

U. S. Fish and Wildlife Service (USFWS). 2007d. Endangered and Threatened Wildlife and Plants; Removing the Bald Eagle in the Lower 48 States From the List of Endangered and Threatened Wildlife. Federal Register 72(130):37346-37372.

U. S. Fish and Wildlife Service (USFWS). 2007e. Endangered and Threatened Wildlife and Plants; Draft Post-Delisting Monitoring Plan for the Bald Eagle (*Haliaeetus leucocephalus*) and Proposed Information Collection. Federal Register 72(130):37373-37374.

U. S. Fish and Wildlife Service (USFWS). 2008. Authorizations Under the Bald and Golden Eagle Protection Act for Take of Eagles. Federal Register 73(98):29075-29084.

U. S. Fish and Wildlife Service (USFWS). 2009a. Endangered and Threatened Wildlife and Plants; Status Review of the Bald Eagle (*Haliaeetus leucocephalus*) in the Sonoran Desert Area of Central Arizona and Northwestern Mexico. Federal Register 74(10):2465-2467.

U. S. Fish and Wildlife Service (USFWS). 2009b. Eagle Permits; Take Necessary To Protect Interests in Particular Localities (final rule). Federal Register 74(175):46836-46879.

Watson, J. W., M. G. Garrett, and R. G. Anthony. 1991. Foraging ecology of bald eagles in the Columbia River estuary. Journal of Wildlife Management 55(3):492-499.

California Spotted Owl

Beck, T. W., and G. I. Gould, Jr. 1992. Background and the current management situation for the California spotted owl. Pp 37-53 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck (technical coordinators); The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Berigan, W.J., R.J Gutièrrez, D.J. Tempel. 2012. Evaluating the efficacy of protected habitat areas for the California spotted owl using long-term monitoring data. Wildlife Management 110(6): 299-303.

Bias, M. A., and R. J. Gutiérrez. 1992. Habitat associations of California spotted owls in the central Sierra Nevada. Journal of Wildlife Management 56(3):584-595.

Bingham, B. B., and B. R. Noon. 1997. Mitigation of habitat “take”: application to habitat to conservation planning. Conservation Biology 11(1):127-139.

Blakesley, J. A., B. R. Noon, and D. W. H. Shaw. 2001. Demography of the California spotted owl in northeastern California. The Condor 103:667-677.

Blakesley, J. A. 2003. Ecology of the California spotted owl: Breeding dispersal and associations with forest stand characteristics in northeastern California. Dissertation, Colorado State University, Fort Collins, Colorado.

Blakesley, J. A., B. R. Noon, and D. R. Anderson. 2005. Site occupancy, apparent survival, and reproduction of California spotted owls in relation to forest stand characteristics. Journal of Wildlife Management 69(4):1554-1564.

Blakesley, J. A., M. E. Seamans, M. M. Conner, A. B. Franklin, G. C. White, R. J. Gutiérrez, J. E. Hines, J. D. Nichols, T. E. Munton, D. W. H. Shaw, J. J. Keane, G. N. Steger, B. R. Noon, T. L. McDonald, and S. Britting. 2006a. Demography of the California spotted owl in the Sierra Nevada: Report to the U.S. Fish and Wildlife Service on the January 2006 Meta-Analysis.

Blakesley, J. A., D. R. Anderson, and B. R. Noon. 2006b. Breeding dispersal in the California spotted owl. Condor 108:71-81.

Bond, M. L., M. E. Seasmans, and R. J. Gutiérrez. 2004. Modeling nesting habitat selection of California spotted owls (*Strix occidentalis occidentalis*) in the central Sierra Nevada using standard forest inventory metrics. Forest Science 50(6):773-780.

Bond, M. L., D. E. Lee, R. B. Siegel, and J. P. Ward, Jr. 2009. Habitat use and selection by California spotted owls in a postfire landscape. Journal of Wildlife Management 73(3): 1116-1124.

Call, D. R., R. J. Gutiérrez, and J. Verner. 1992. Foraging habitat and home-range characteristics of California spotted owls in the Sierra Nevada. Condor 94:880-888.

Chatfield, A. H. 2005. Habitat Selection by a California Spotted Owl Population: A Landscape Scale Analysis Using Resource Selection Functions. Master's Thesis, Department of Fisheries, Wildlife, and Conservation Biology, University of Minnesota.

Coppeto, S. A., D. A. Kelt, D. H. Van Vuren, J. A. Wilson, and S. Bigelow. 2006. Habitat associations of small mammals at two spatial scales in the northern Sierra Nevada. Journal of Mammalogy 87(2):402-413.

Gutiérrez, R. J., J. Verner, K. S. McKelvey, B. R. Noon, G. N. Steger, D. R. Call, W. S. LaHaye, B. B. Bingham, and J. S. Senser. 1992. Habitat Relations of the California Spotted Owl. Pages 79-98 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck (technical coordinators); The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Gutiérrez, R. J., and G. F. Barrowclough. 2005. Redefining the distributional boundaries of the northern and California spotted owls: implications for conservation. The Condor 107:182-187.

Hayward, L. S., A. Bowles, J. C. Ha, and S. K. Wasser. 2011. Impacts of acute and long-term vehicle exposure on physiology and reproductive success of the northern spotted owl. Ecosphere 2(6):art65. doi:10.1890/ES10-00199.1

Hunsaker, C. T., B. B. Boroski, and G. N. Steger. 2002. Relations between canopy cover and the occurrence and productivity of California spotted owls *in* J. M. Scott, P. J. Hegland, and M. L. Morrison (Eds.); Predicting Species Occurrences: Issues of Accuracy and Scale. Island Press, Washington, DC, pp. 687-700.

Innes, R. J., D. H. Van Vuren, D. A. Kelt, M. L. Johnson, J. A. Wilson, and P. A. Stine. 2007. Habitat associations of dusky-footed woodrats (*Neotoma fuscipes*) in mixed-conifer forest of the northern Sierra Nevada. Journal of Mammology 88(6):1523-1531.

Innes, R. J., D. H. Van Vuren, and D. A. Kelt. 2008. Characteristics and use of tree houses by dusky-footed woodrats in the northern Sierra Nevada. Northwestern Naturalist 89:109-112.

Irwin, L. L., L. A. Clark, D. C. Rock, and S. L. Rock. 2007. Modeling Foraging Habitat of California Spotted Owls. Journal of Wildlife Management 71(4):1183-1191.

Irwin, L.L., D. R. Rock, S.C. Rock, C. Loehle, and P. Van Deusen. 2015. Forest ecosystem restoration: Initial response of spotted owls to partial harvesting. Forest and Ecology Management 354 (2015) 232-242.

Ishak, H. D., J. P. Dumbacher, N. L. Anderson, J. J. Keane, G. Valkiūnas, S. M. Haig, L. A. Tell, and R. N. M. Sehgal. 2008. Blood parasites in owls with conservation implications for the spotted owl (*Strix occidentalis*). PLoS ONE 3(5): e2304. Doi:10.1371/journal.pone.0002304.

Jones, G.M., J.J. Keane, R.J. Gutiérrez, and M.Z. Peery. 2017. Declining old-forest species as a legacy of large trees lost. Divers Distrib. 2017: 00:1-11 https://doi.org/10.1111/ddi.12682

Jones, G. M., R. J. Guti´errez, D. J. Tempel, S. A. Whitmore, W. J.,Berigan, and M. Z. Peery (2016a). Megafires: An emerging threat to old-forest species. Frontiers in Ecology and the Environment 14:300–306.

LaHaye, W. S., R. J. Gutiérrez, and D. R. Call. 1997. Nest-site selection and reproductive success of California spotted owls. Wilson Bulletin 109(1):42-51.

LaHaye, W. S., G. S. Zimmerman, and R. J. Gutiérrez. 2004. Temporal variation in the vital rates of an insular population of spotted owls (*Strix occidentalis occidentalis*): Contrasting effects of weather. The Auk 121(4):1056-1069.

Lande, R. 1988. Demographic models of the northern spotted owl. *Oecologia* 75: 601-.607.

Lee, D. C., and L. L. Irwin. 2005. Assessing risks to spotted owls from forest thinning in fire-adapted forests of the western United States. Forest Ecology and Management 211:191-209.

Lee, D. E., and W. D. Tietje. 2005. Dusky-footed woodrat demography and prescribed fire in a California oak woodland. Journal of Wildlife Management 69(2):1211-1220.

Livezey, K. B. 2007. Barred owl habitat and prey: a review and synthesis of the literature. Journal of Raptor Research 41(3):177-201.

Livezey, K. B., M. F. Elderkin, P. A. Cott, J. Hobbs, and J. P. Hudson. 2008a. Barred owls eating worms and slugs: the advantage in not being picky eaters. Northwestern Naturalist 89:185-190.

Livezey, K. B. 2008b. Range expansion of barred owls, part I: chronology and distribution. American Midland Naturalist 161:49-56.

McEachern, M. B., C. A. Eagles-Smith, C. M. Efferson, and D. H. Van Vuren. 2006. Evidence for local specialization in a generalist mammalian herbivore, *Neotoma fuscipes*. Oikos 113:440-448.

Meyer, M. D., D. A. Kelt, and M. P. North. 2005a. Nest trees of northern flying squirrels in the Sierra Nevada. Journal of Mammology 86(2):275-280.

Meyer, M. D., and M. P. North. 2005. Truffle abundance in riparian and upland mixed-conifer forest of California’s southern Sierra Nevada. Canadian Journal of Botany 83:1015-1020.

Meyer, M. D., M. P. North, and D. A. Kelt. 2007a. Nest trees of northern flying squirrels in Yosemite National Park, California. The Southwestern Naturalist 52(1): 157-161.

Meyer, M. D., D. A. Kelt, and M. P. North. 2007b. Microhabitat associations of northern flying squirrels in burned and thinned forest stands of the Sierra Nevada. American Midland Naturalist 157:202-211.

Moen, C. A. and R. J. Gutiérrez. 1997. California spotted owl habitat selection in the central Sierra Nevada. Journal of Wildlife Management 61(4):1281-1287.

Munton, T. E., K. D. Johnson, G. N. Steger, and G. P. Eberlein. 2002. Diets of California spotted owls in the Sierra National Forest. Pp 99-105 *in* J. Verner (technical editor); Proceedings of a Symposium on the Kings River Sustainable Forest Ecosystem Project: Progress and Current Status. Gen. Tech. Rep. PSW-GTR-183. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Noon, B. R., and C.M. Biles. 1990. Mathematical demography of spotted owls in the Pacific Northwest. Journal of Wildlife Management 54(1): 18-27.

North, M., J. T. Kane, V. R. Kane, G. P. Asner, W. Berigan, D.J. Churchill, S. Conway, R.J. Gutierrez, S. Jeronimo, J. Keane, A. Koltunov, T. Mark, M. Moskal, T. Munton, Z. Peery, C. Ramirez, R. Sollmann, A. M. White, and S. Whitmore. 2017. Cover of tall trees best predicts California spotted owl habitat. Forest Ecology and Management 405 (2017) 166-178.

North, M., G. Steger, R. Denton, G. Eberlein, T. Munton, and K. Johnson. 2000. Association of weather and nest-site structure with reproductive success in California spotted owls. Journal of Wildlife Management 64(3):797-807.

Seamans, M. E. 2005. Population biology of the California spotted owl in the central Sierra Nevada. Dissertation, University of Minnesota, St. Paul, Minnesota.

Stephens, S. L., J. D. Miller, B. M. Collins, M. P. North, J. J. Keane, and S. L. Roberts. 2016. Wildfire impacts on California spotted owl nesting habitat in the Sierra Nevada. Ecosphere 7(11):e01478. 10.1002/ecs2.1478

Tempel, D.J., J.J. Keane, R.J. Gutierrez, J.D. Wolfe, G. M. Jones, A. Koltunov, C.M. Ramirez, W.J. Berigan, C.V. Gallagher, T.E. Munton, P.A. Shaklee, S.A. Whitmore, and M.Z. Peery. 2016. Meta-analysis of California Spotted Owl (*strix occidentalis occidentalis*) territory occupancy in the Sierra Nevada: Habitat associations and their implecations for forest management. The Condor 118: 747-765.

Tempel, D.J., R.J. Gutierrez, J.J. Battles, D.L. Fry, Y. Su, Q. Guo, M.J. Reetz, S.A. Whitmore, G.M. Jones, B.M. Collins, S.L. Stephens, M. Kelly, W.J. Berigan, and M.Z. Peery. 2015. Evaluating short and long term impacts of fuels treatments and simulated wildfire on an old-forest species. Ecosphere 6(12): 1-18.

Tempel, D.J. 2014. California spotted owl dynamics in the central Sierra Nevada: An assessment using multiple types of data. Dissertation, University of Minnesota.

Tempel, D.J., R.J. Gutierrez, S.A Whitmore, M.J. Reetz, R.E. Stoelting, W.J. Berigan, M.E. Seamans, and M.Z. Peery. 2014. Effects of forest management on California spotted owls: implications for reducing wildfire risk in fire-prone forests. Ecological Applications 24(8): 2089-2106.

USDA Forest Service. 1993. Protocol for surveying for spotted owls in proposed management activity areas and habitat conservation areas.

USDA Forest Service. 2009. California spotted owl module: 2008 annual report (on the Plumas-Lassen study). Keane, J.J. (Principal Investigator), C.V. Gallagher, R.A. Gerrard, G. Jehle, and P.A. Shaklee. USDA Forest Service Pacific Southwest Research Station, Davis, California. 36 pp.

U. S. Fish and Wildlife Service (USFWS). 2003. Endangered and Threatened Wildlife and Plants: 12-Month Finding for a Petition to List the California Spotted Owl (*Strix occidentalis occidentalis*). Federal Register 68(31):7580-7608.

U. S. Fish and Wildlife Service (USFWS). 2006. Endangered and Threatened Wildlife and Plants: 12-Month Finding for a Petition to List the California Spotted Owl (*Strix occidentalis occidentalis*). Federal Register 71(100):29886-29906.

U.S. Fish and Wildlife Service (USFWS). 2015. Endangered and Threatened Wildlife and plants; 90 day findings on 25 petitions: Federal Register 80 (181): 29886.

Verner, J., K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck. 1992a. Assessment of the current status of the California spotted owl, with recommendations for management. Pp 3-26 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck (technical coordinators); The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Verner, J., K. S., R. J. Gutiérrez, and G. I. Gould, Jr. 1992b. The California spotted owl: General biology and ecological relations. Pp 55-77 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck (technical coordinators); The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Wasser, S. K., K. Bevis, G. King and E. Hanson. 1997. Noninvasive physiological measures of disturbance in the northern spotted owl. Conservation Biology 4:1019-1022.

Williams, D. F., J. Verner, H. F. Sakai, and J. R. Waters. 1992. General biology of major prey species of the California spotted owl. Pp 207-221 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck (technical coordinators); The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Zabel, C. J., G. N. Steger, K. S. McKelvey, G. P. Eberlein, B. R. Noon, and J. Verner. 1992. Home-range size and habitat-use patterns of California spotted owls in the Sierra Nevada. Pp 149-163 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, Jr., and T. W. Beck (technical coordinators); The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California.

Zabel, C. J., K. McKelvey, and J. P. Ward, Jr. 1995. Influence of primary prey on home-range size and habitat-use patterns of northern spotted owls (*Strix occidentalis caurina*). Canadian Journal of Zoology 73: 433-439.

Great Gray Owl

Beck, T. W. 1985. Interim direction for management of great gray owl. USDA Forest Service, Stanislaus National Forest. Sonora, CA. 24pp.

Brunton, D. F. 1971. Observations of the great gray owl on winter range. Can. Field Nat. 86: 315-322.

Bull, E. L., M. G. Henjum, and R. S. Rohweder. 1989. Diet and optimal foraging of great gray owls. Journal of Wildlife Management 53(1):47-50.

Bull, E. L. and M. G. Henjum. 1990. Ecology of the great gray owl. General Technical Report PNW-GTR 265. U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon. 39pp.

Greene, C. 1995. Habitat requirements of great gray owls in the central Sierra Nevada. M.S. Thesis. University of Michigan. 94pp.

Hull, J.M., A. Engilis, JR., J.R. Medley, E.P. Jepsen, J.R. Duncan, H.B. Ernest, and J.J. Keane. 2014. A new subspecies of Great Gray Owl (strix nebulosi) in the Sierra Nevada of California, U.S.A. Journal of Raptor Reasearch 48: 68-77.

Jones, C. A., and C. N. Baxter. 2004. *Thomomys bottae*. Mammalian Species 742:1-14.

Polasik, J. S., J. X. Wu, K. N. Roberts, and R. B. Siegel. 2016. Great Gray Owls nesting in atypical, low-elevation habitat in the Sierra Nevada, California. Journal of Raptor Research, 50(2): 194-206.

Quady, D. E. 2008. Featured photo: great gray owls nesting in Fresno County, California. Western Birds 39:110-116.

Reid, M. E. 1989. The predator-prey relationships of the great gray owl in Yosemite National Park. Cooperative National Park Research Studies Unit Technical Report No. 35. 86pp.

Rognan, C. B. 2007. Bioacoustic techniques to monitor great gray owls (Strix nebulosa) in the Sierra Nevada. M. A. Thesis, Humboldt State University, CA. 64 pp.

Sears, C. L. 2006. Assessing distribution, habitat suitability, and site occupancy of great gray owls (*Strix nebulosa*) in Calfornia. M. S. Thesis, University of California, Davis. 81 pp.

Sera, W. E., and C. N. Early. 2003. *Microtus montanus*. Mammalian Species 716:1-10.

USDA Forest Service. 2000. Survey Protocol for the Great Gray Owl in the Sierra Nevada of California. 32pp.

Van Riper, C., III, and J. van Wagtendonk. 2006. Home range characteristics of great gray owls in Yosemite National Park, California. Journal of Raptor Research 40(2):130-141.

Wilson, J. 1981. Some aspects of the ecology of the great gray owl in the Central Sierra Nevada. USDA Forest Service. 30pp.

Wilson, J. 1985. Great gray owl survey, 1984. California Department of Fish and Game. Project W-65-R-2. Progress Report.

Winter, J. 1981. Some aspects of the ecology of the great gray owl in the Central Sierra Nevada. USDA, Forest Service. Stanislaus National Forest Contract # 43-2276. Final report. Sonora, CA. 30pp.

Winter, J. 1982. Further investigations on the ecology of the great gray owl in the central Sierra Nevada. Final Report to the Forest Service under contract #43-2348. Stanislaus National Forest, Sonora CA. 35pp.

Winter, J. 1986. Status, distribution and ecology of the great gray owl (*Strix nebulosa*) in California. San Francisco, CA: San Francisco State University. 121pp. M.S. thesis.

Wu, J.X., R.B. Siegel, H.L. Loffland, M.W. Tingley, S.L. Stock, K.N. Roberts, J.J. Keane, J.R. Medley, R. Bridgman, and C. Stermer.

Northern Goshawk

Andersen, D. E., S. DeStefano, M. I. Goldstein, K. Titus, C. Crocker-Bedford, J. J. Keane, R. G. Anthony, and R. N. Rosenfield. 2005. Technical review of the status of northern goshawks in the western United States. Journal of Raptor Research 39(3):192-209.

Austin, K. A. 1991. Habitat use of northern goshawk in Southern Cascades. M.S. Thesis. Oregon State University, Corvallis, Oregon. In prep.

Beier, P. and J. E. Drennan. 1997. Forest structure and prey abundance in foraging areas of northern goshawks. Ecological Applications 7:564-571.

Bloom, P. H., G. R. Stewart and B. J. Walton. 1986. The status of northern goshawk in California, 1981-1983. Adm. Rept. 85-1. State of California, Department of Fish and Game, Sacramento, CA.

Daw, S. K., and S. DeStefano. 2001. Forest characteristics of northern goshawk nest stands and post-fledging areas in Oregon. Journal of Wildlife Management 65(1):59-65.

Fowler, C. 1988. Habitat capability model for the northern goshawk. USFS, Tahoe National Forest, Nevada City, CA. 21 pp.

Grubb, T. G., L. L. Pater, and D. K. Delaney. 1998. Logging Truck Noise Near Nesting Northern Goshawks. USDA Forest Service, Rocky Mountain Research Station. Research Note RMRS-RN-3. 2 pp.

Hargis, C. D., R. D. Perloff, and C. McCarthy. 1991. Home ranges and habitats of northern goshawks in eastern California. Draft Rep. C.D. Hargis, Dept. Fish and Wildl., Utah State Univ. Logan, UT. 17 pp.

Hayward, G. D., and R. E. Escano. 1989. Goshawk nest-site characteristics in western Montana and northern Idaho. The Condor 91:476-479.

Kennedy, P. L. 2003. Northern Goshawk (*Accipiter gentilis atricapillus*): A technical conservation assessment. USDA Forest Service, Rocky Mountain Region.

Morrison, M. L., R. J. Young, J. S. Romsos, and R. Golightly. 2011. Restoring forest raptors: influence of human disturbance and forest condition on northern goshawks. Restoration Ecology 19(2):273-279.

Moser, B. W., and E. O. Garton. 2009. Short-term effects of timber harvest and weather on northern goshawk reproduction in northern Idaho. The Journal of Raptor Research 43(1):1-10.

Squires, J. R., and L. F. Ruggiero. 1996. Nest-site preference of northern goshawks in southcentral Wyoming. Journal of Wildlife Management 60(1):170-177.

Reynolds, R. T., R. T. Graham, M. H. Reiser, R. L. Bassett, P. L. Kennedy, D. A. Boyce, Jr., G. Goodwin, R. Smith, and E. L. Fisher. 1992. Management recommendations for the northern goshawk in the southwestern United States. Gen. Tech. Report RM-217. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, 90 pp.

USDA Forest Service. 1999. New information and management considerations for the northern goshawk (*Accipter gentilis*). Tahoe National Forest, Nevada City, CA. 16 pp.

USDA Forest Service. 2000. Survey Protocol for the Northern Goshawk in the Pacific Southwest Region.

U. S. Fish and Wildlife Service (USFWS). 1998. Endangered and Threatened Wildlife and Plants; Notice of 12-Month Finding on a Petition To List the Northern Goshawk in the Contiguous United States West of the 100^{th} Meridian. Federal Register 63(124):35183-35184.

Verner, J. and A.S. Boss, technical coordinators. 1980. California wildlife and their habitats: Western Sierra Nevada. USDA Forest Service, PSW-37., Berkeley, CA. 439 pp.

Willow Flycatcher

Bombay, H. L. 1999. Scale perspectives in habitat selection and reproductive success for willow flycatchers *(Empidonax traillii*) in the centeral Sierra Nevada, California. Masters thesis, California State University, Sacramento. 223 pp.

Bombay, H. L. 2003. Annual report and preliminary demographic analysis for willow flycatcher monitoring in the Central Sierra Nevada in partial fulfillment of contract 53-91S8-01-NRM09 between Dr. Michael L. Morrison under subcontract to Tetra Tech FW, Inc. and USDA Forest Service, Pacific Southwest Region, Vallejo, CA. 50 pp.

Bombay, H. L., M. L. Morrison, D. E. Taylor, and J. W. Cain III. 2001. Annual report and preliminary demographic analysis for willow flycatcher monitoring in the Central Sierra Nevada in partial fulfillment of contract RFQ-IBET-17-01-053 between University of California, San Diego - White Mountain Research Station and USDA Forest Service, Tahoe National Forest. 46 pp.

Bombay, H. L., M. L. Morrison, and L. S. Hall. 2003. Scale perspectives in habitat selection and animal performance for willow flycatchers (*Empidonax traillii*) in the central Sierra Nevada, CA. Studies in Avian biology No. 26:60-72, 2003.

Cain, J. W., III, M. L. Morrison, and H. L. Bombay. 2003. Predator activity and nest success of willow flycatchers and yellow warblers. Journal of Wildlife Management 67(3):600-610.

Cain, J. W., III., K. S. Smallwood, M. L. Morrison, and H. L. Loffland. 2006. Influence of mammal activity on nesting success of passerines. Journal of Wildlife Management 70(2):522-531.

Green, G. A., H. L. Bombay, M. L. Morrison. 2003. Conservation assessment of the willow flycatcher in the Sierra Nevada. USDA Forest Service, Pacific Southwest Region, Vallejo, CA. 62 pp.

Fowler, C., B. Valentine, S. Sanders and M. Stafford. 1991. Habitat Suitability Index Model: willow flycatcher (*Empidonax traillii*). Unpublished document, USDA Forest Service, Tahoe National Forest. 15 pp.

Harris, J. H., S. D. Sanders, and M. A. Flett. 1987. Willow flycatcher surveys in the Sierra Nevada. Western Birds 18:27-36.

Harris, J. H., S. D. Sanders, and M. A. Flett. 1988. The status and distribution of the willow flycatcher in the Sierra Nevada: results of the 1986 Survey. California Department of Fish and Game. Wildlife Management Division Administrative Report. 88-1. 32 pp.

Loffland, H L., and R.B. Siegel. 2014. 2014 Willow Flycatcher surveys in east-side meadows on the Tahoe National Forest. The Institute for Bird Populations, Point Reyes Station, CA.

Loffland, H.L., R.B. Siegel, C. Stermer, B.R. Campos, R.D. Burnett, and T. Mark. 2014. Assessing Willow Flycatcher population size and distribution to inform meadow restoration in the Sierra Nevada and Southern Cascades. The Institute for Bird Populations, Point Reyes Station, CA.

Mathewson, H.A., M.L. Morrison, H.L. Loffland, and P.F. Brussard. 2012. Ecology of willow flycatchers (*Empidonax traillii*) in the Sierra Nevada, California: effects of meadow characteristics and weather on demographics. Ornithological Monographs 75:1-31.

Mathewson, H. A., H. L. Loffland, M. L. Morrison, L. Vormwald, and C. Cocimano. 2009. 2008 annual report and preliminary demographic analysis for willow flycatcher monitoring in the central Sierra Nevada. Produced under Cost Share Agreement 06-CR-11052007-160 between Texas A&M University and USDA Forest Service, Region 5.

Mayfield, H. 1961. Nesting success calculated from exposure. The Wilson Bulletin 73(3):255-261.

McCreedy, C., and S. K. Heath. Atypical willow flycatcher nesting sites in a recovering riparian corridor at Mono Lake, California. Western Birds 35:197-209.

Morrison, M. L., H. L. Bombay, J. W. Cain III, and D. E. Taylor. 2000. Annual Report and Preliminary Demographic analysis for Willow Flycatcher monitoring in the Central Sierra Nevada. Unpublished document, USDA Forest Service, Tahoe National Forest. 34 pp.

Powers, L. 1993. Summary of monitoring efforts for a population of willow flycatchers in Perazzo Meadows 1992 and 1993. 4pp.

Sanders, S. D., and M. A. Flett. 1989. Ecology of a Sierra Nevada population of willow flycatchers (*Empidonax traillii*), 1986-1987. Wildlife Management Division Administrative Report 88-3. 34pp.

Serena, M. 1982. The status and distribution of the willow flycatcher (*Empidonax traillii*) in selected portions of the Sierra Nevada, 1982. Calif. Dept. of Fish and Game, Wildlife Management Division, Administrative Report No. 82-5. 28 pp.

Steele, J. 1993. Preliminary report of neotropical migratory bird monitoring at San Francisco State University's Sierra Nevada Field Campus for the Summers of 1992 and 1993. 8 pp.

Strohm, K.H., and H.L. Loffland. 2015. 2015 Willow Flycatcher Surveys in the Tahoe National Forest. The Institute for Bird Populations, Point Reyes Station, CA.

USDA Forest Service. 1994. Biological Evaluation for Managing willow flycatcher (*Empidonax traillii*) Habitat on Plumas National Forest Range Allotments. DRAFT. 26 pp.

USDA Forest Service. 2000. A Willow Flycatcher Survey Protocol for California. 48 pp.

U. S. Fish and Wildlife Service (USFWS). 1995. Endangered and Threatened Wildlife and Plants; Final Rule Determining Endangered Status for the Southwestern Willow Flycatcher. Federal Register 60(38):10694-10715.

Greater Sandhill Crane

California Department of Fish and Game. 1994. Five-year status review: greater sandhill crane (*Grus canadensis tabida*). Reported to: California Fish and Game Commission. 12 pp.

Grinnell, J. and A. H. Miller. 1994. The distribution of the birds of California. Pacific Coast Avifauna, No. 27. 608 pp.

James, A. H. 1997. Sandhill cranes breeding in Sierra Valley, Ca. West Birds 8:159-160.

Johnsgard, P. A. 1975. North American game birds of upland and shoreline. Univ. Nebraska Press, Lincoln. 183 pp.

Harrison, C. 1978. A field guide to the nests, eggs and nestlings of North American birds. W. Collins Sons and Co., Cleveland OH. 416 pp.

Ivey, G. 1995. Draft paper given at the Sandhill Crane Workshop, August 8-9, 1995. Preliminary unpublished data. Alturas, CA.

Littlefield, C. D. 1982. The status and distribution of greater sandhill cranes in California, 1981. The California Resources Agency, Department of Fish and Game.

Littlefield, C. D. 1989. Status of greater sandhill crane breeding populations in California, 1988. State of Calif. the Resources Agency, Dept. of Fish and Game, Wildlife Management Division, Nongame Bird and Mammal Section. 40 pp.

Miller, R. S., G. S. Hochbaum and D. B. Botkin. 1972. A simulation model for the management of sandhill cranes. Yale University School of Forestry and Environmental Studies Bulletin 80.

Pacific Flyway Council. 1997. Pacific Flyway management plan for the Central Valley population of greater sandhill cranes. Pacific Flyway Study Comm. [c/o Pacific Flyway Representative USFWS], Portland, OR 97232-4181. Unpubl. Rept. 44pp. and appendices.

Schlorff, R. 1998. Wildlife Biologist, Sensitive Species Branch, California Dept. of Fish and Game, Sacramento, CA. pers. comm.

USDA Forest Service. 1994. Management guidelines for the greater sandhill crane on National Forest Systems Lands in California. Pacific Southwest Region, San Francisco, Ca. 40 pp.

Youngblood, Q. 1998. District Biologist, Tahoe National Forest. personal communication.

Zeiner, D. C., W. F. Laudenslayer, Jr., K. E. Mayer and M. White, (Eds.). 1990. California wildlife: Volume II: Birds. State of California, The Resources Agency, Dept. of Fish and Game, Sacramento, CA pp. 184-185.

Pacific Marten and California Wolverine

Allen, A. W. 1987. The relationship between habitat and furbearers. Pages 164-179 *in* M. Novak, J.A. Baker, M.E. Obbard and B. Malloch (editors); Wild furbearer management and conservation in North America. Ontario Ministry of Natural Resources, Canada. 1150 pp.

Andruskiw, M., J. M. Fryxell, I. D. Thompson, and J. A. Baker. 2008. Habitat-mediated variation in predation risk by the American marten. Ecology 89(8):2273-2280.

Aubry, K. B., K. S. McKelvey, and J. P. Copeland. 2007. Distribution and broadscale habitat relations of the wolverine in the contiguous United States. Journal of Wildlife Management 71(7):2147-2158.

Aubry, Keith B., William J. Zielinski, Martin G. Raphael, Gilbert Proulx, and Steven W. Buskirk, eds. 2012. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis.

Bateman, M. C. 1986. Winter habitat use, food habits and home range size of the marten, *Martes americana*, in western Newfoundland. Canadian Field Naturalist 100: 58-62.

Bennett, L. A. and F. B. Samson. 1984. Marten ecology and habitat management in the central Rocky Mountains: problem analysis. USDA Forest Service and Colorado Cooperative Wildlife Research Unit, Colorado State University. Fort Collins, Colorado. 60 pp.

Buck, S., C. Mullis and A. Mossman. 1983. Final report: Coral Bottom - Hayfork Bally fisher study. USDA Forest Service. Unpublished report. 133 pp.

Bull, E. L., and T. W. Heater. 2000. Resting and denning sites of American martens in northeastern Oregon. Northwest Science 74(3):179-185.

Bull, E. L., and T. W. Heater. 2001. Home range and dispersal of the American marten in northeastern Oregon. Northwestern Naturalist 82:7-11.

Bull, E. L., T. W. Heater, and J. F. Shepard. 2005. Habitat selection by the American marten in northeastern Oregon. Northwest Science 79(1):37-43.

Buskirk, S. W., S. C. Forrest, M. G. Raphael, and H. J. Harlow. 1989. Winter resting site ecology of marten in the central Rocky Mountains. Journal of Wildlife Management 53(1):191-196.

Buskirk, S. W. and L. F. Ruggiero. 1994. American marten. Pages 7-37 *in* L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski (technical editors); The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. Gen. Tech. Rep. RM-254. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Buskirk, S. W., and W. J. Zielinski. 1997. American marten (*Martes americana*) ecology and conservation. Pages 17-22 *in* J.E. Harris and C.V. Ogan (editors). Mesocarnivores of northern California: biology, management, and survey techniques, workshop manual. August 12-15, 1997. Humboldt State University, Arcata, California. The Wildlife Society, California North Coast Chapter, Arcata, California. 127 pp.

Butts, T. W. 1992. Wolverine (*Gulo gulo*) biology and management: A literature review and annotated bibliography. Unpublished paper for the U.S. Forest Service. Northern Region.

Carroll, C., R. F. Noss, and P. C. Paquet. 2001. Carnivores as focal species for conservation planning in the Rocky Mountain region. Ecological Applications 11(4):961-980.

Chapel, M., A. Carlson, D. Craig, T. Flaherty, C. Marshall, M. Reynolds, D. Pratt, L. Pyshora, S. Tanguay and W. Thompson. 1992. Recommendations for managing late-seral-stage forest and riparian habitats on the Tahoe National Forest. Located at: USDA Forest Service, Tahoe National Forest, Nevada City, California. 31pp.

Chapin, T. G., D. J. Harrison, D. M. Phillips. 1997. Seasonal habitat selection by marten in an untrapped forest preserve. Journal of Wildlife Management 61:707-717.

Chapin, T. G., D. J. Harrison, and D. D. Katnik. 1998. Influence of landscape pattern on habitat use by American marten in an industrial forest. Conservation Biology 12(6):1327-1337.

Clark, W. R. and E. K. Fritzell. 1992. A review of population dynamics of furbearers. In: Wildlife 2001: populations. D. R. McCullough and R. H. Barrett, eds. Elsevier Applied Science, London. 1163 pp.

Copeland, J. P., and T. E. Kucera. 1997. Wolverine (*Gulo gulo*). Pages 23-33 *in* J. E. Harris and C. V. Ogan (editors). Mesocarnivores of northern California: biology, management, and survey techniques, workshop manual. August 12-15, 1997. Humboldt State University, Arcata, California. The Wildlife Society, California North Coast Chapter, Arcata, California. 127 pp.

Copeland, J. P., J. M. Peek, C. R. Groves, W. E. Melquist, K. S. McKelvey, G. W. McDaniel, C. D. Long, and C. E. Harris. 2007. Seasonal habitat associations of the wolverine in central Idaho. Journal of Wildlife Management 71(7):2201-2212.

Dalerum, F., J. Loxterman, B. Shults, K. Kunkel, and J.A. Cook. 2007. Sex-specific dispersal patterns of wolverines: insights from microsatellite markers. Journal of Mammalogy 88(3):793-800.

Davis: University of California, Centers for Water and Wildland Resources. 1996. Sierra Nevada Ecosystem Project, Final Report to Congress.

De Vos, A. 1951. Recent findings in fisher and marten ecology and management. Transactions North American Wildl. Conf. 16:498-505.

Drew, G. S. 1995. Winter habitat selection by American marten (Martes americana) in Newfoundland: why old growth? Dissertation, Utah State University, Logan, Utah. 100 pp.

Dumyahn, J. B., P. A. Zollner, and J. H. Gilbert. 2007. Winter home-range characteristics of American marten (*Martes americana*) in northern Wisconsin. American Midland Naturalist 158:382-394.

Duncan Furbearer Interagency Workgroup. 1989. Workgroup assembled to review the proposed Duncan Timber Sale, Tahoe National Forest and formulate proposed Management Guidelines. Members present: Slader Buck, Reginald Barrett, Terri Simon-Jackson, Gordon Gould, Ron Schlorff, Jeff Finn, Joelle Buffa, Maeton Freel, Jeff Mattison, Mike Chapel, Mariana Armijo, Julie Lydick and Phil Turner.

Freel, M. 1991. A literature review for management of fisher and marten in California. Unpubl. Document, USDA Forest Service, Pacific Southwest Region.

Fry, W. 1923. The wolverine. California Fish and Game 9(4):129-134.

Fuller, A. K., and D. J. Harrison. 2005. Influence of partial timber harvesting on American martens in north-central Maine. Journal of Wildlife Management 69(2):710-722.

Fuller, A. K. 2006. Multi-scalar responses of forest carnivores to habitat and spatial pattern: case studies with Canada lynx and American martens. Dissertation, University of Maine, Orono, Maine.

Golden, H. N., J. D. Henry, E. F. Becker, M. I. Goldstein, J. M. Morton, D. Frost, Sr., and A. J. Poe. 2007. Estimating wolverine Gulo gulo population size using quadrat sampling of tracks in snow. Wildlife Biology 13(Suppl. 2):52-61.

Grinnell, J. 1913. (*as cited* in Shempf and White 1977) A distributional list of the mammals of California. Calif. Acad. Sci. Proc. Ser. 4, 3:265-390.

Grinnell, J. 1933. (*as cited* in Shempf and White 1977) Review of the recent mammal fauna of California. Univ. Calif. Publ. Zool. 40(2):71-234.

Grinnell, J., J. S. Dixon, and J.M. Linsdale. 1937. Fur-bearing mammals of California. University of California Press, Berkeley, California.

Hargis, C. D. and J. A. Bissonette. 1995. The effect of forest fragmentation on American marten populations and prey availability. Final report for the Utah Division of Wildlife Resources, Wasatch-Cache National Forest, Ashley National Forest, Utah Wilderness Association (Contract No. 91-9166). December 31, 1995. 96pp.

Hargis, C. D. and D. R. McCullough. 1984. Winter diet and habitat selection of marten in Yosemite National Park. Journal of Wildlife Management 48(1):140-146.

Hargis, C. D. and J. A. Bissonette. 1997. The influence of forest fragmentation and prey availability on American marten populations: a landscape level analysis. Pages 437-451 *in* G. Proulx, H. N. Bryant and P. M. Woodard (editors). *Martes*: taxonomy, ecology, techniques, and management. Proceedings of the Second International *Martes* symposium. The Provincial Museum of Alberta. Edmonton, Alberta, Canada. 474 p.

Hargis C. D., J. A. Bissonette, and D. L. Turner. 1999. The influence of forest fragmentation and landscape pattern on American martens. Journal of Applied Ecology 36:157-172.

Hedmark, E., J. Persson, P. Segerström, A. Landa, and H. Ellegren. 2007. Paternity and mating system in wolverines Gulo gulo. Wildlife Biology 13(Suppl. 2):13-30.

Hornocker, M. G. and H. S. Hash. 1981. Ecology of the wolverine in northwestern Montana. Canadian Journal of Zoology 59:1286-1301.

Jones, F. L. 1950. Recent records of the wolverine (*Gulo luscus luteus*) in California. California Fish and Game 36:320-322.

Jones, F. L. 1955. Records of southern wolverine, *Gulo luscus luteus*, in California. Journal of Mammalogy 36(4):569.

Kirk, T. A. 2007. Landscape-scale habitat associations of the American marten (*Martes americana*) in the greater southern Cascades region of California. M.S. Thesis, Humboldt State University, Arcata, California, 103 pp.

Koehler, G. M., W. R. Moore and A. R. Taylor. 1975. Preserving the pine marten: management guidelines for western forests. Western Wildlands 2:31-36.

Kovach, S. D. 1981. Wolverine, *Gulo gulo*, records for the White Mountains, California. California Fish and Game 67(2):132-133.

Krebs, J., E. C. Lofroth, and I. Parfitt. 2007. Multiscale habitat use by wolverines in British Columbia, Canada. Journal of Wildlife Management 71(7):2180-2192.

Krohn, W. B., Arthur, S. M., Paragi, T. F. 1994. Mortality and vulnerability of a heavily trapped fisher population. Pp 137-148 *in* S.W. Buskirk, Harestad, A.S., Raphael, M.G., comps., eds. Martens, Sables and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Krohn, W. B., K. D. Elowe, and R. B. Boone. 1995. Relations among fishers, snow, and martens: development and evaluation of two hypotheses. Forestry Chronicle 71(1):97-105.

Krohn, W. B., W. J. Zielinski, and R. B. Boone. 1997. Relations among fishers, snow, and martens in California: results from small-scale spatial comparisons. Pp. 211-232 *in* G. Proulx, H. N. Bryant, and P. M. Woodard (editors); *Martes*: taxonomy, ecology, techniques, and management. Proceedings of the Second International *Martes* Symposium. Provincial Museum of Alberta, Edmonton, Alberta, Canada.

Krohn, W., C. Hoving, D. Harrison, D. Phillips, and H. Frost. 2005. *Martes* foot-loading and snowfall patterns in eastern North America. Pp. 115-131 *in* D. J. Harrison, A. K. Fuller, and G. Proulx (editors); Martens and Fishers (*Martes*) in Human-Altered Environments. Springer-Verlag, New York, New York.

Kucera, T. E., W. J. Zielinski, and R. H. Barrett. 1995. Current distribution of the American marten, *Martes americana*, in California. California Fish and Game 81(3):96-103.

Lamberson, R. H., R. L. Truex, W. J. Zielinski, D. C. Macfarlane. 2000. Preliminary analysis of fisher population viability in the southern Sierra Nevada. Unpublished document, February 15, 2000. Available at U.S. Department of Agriculture, Forest Service, Tahoe National Forest Supervisor’s Office, Nevada City, California. 20pp.

Lofroth, E. C., J. A. Krebs, W. L. Harrower, and D. Lewis. 2007. Food habits of wolverine *Gulo gulo* in montane ecosystems of British Columbia, Canada. Wildlife Biology 13(Suppl. 2):31-37.

Magoun, A. J., and J. P. Copeland. 1998. Characteristics of wolverine reproductive den sites. Journal of Wildlife Management 62(4):1313-1320.

Martin, S. K. 1987. The ecology of the pine marten (*Martes americana*) at Sagehen Creek, California. PhD Thesis. University of California, Berkeley. 223 pp.

Maser, C., J. M. Trappe and D. C. Ure. 1978. Implications of small mammal mycophagy to the management of western coniferous forests. Trans. North Am. Wildl. and Nat. Resour. Conf. 43:78-88.

May, R., A. Landa, J. van Dijk, J.D.C. Linnell, and R. Andersen. 2006. Impact of infrastructure on habitat selection of wolverines *Gulo gulo*. Wildlife Biology 12:285-295.

McDonald, D. W. 1983. The ecology of carnivore social behavior. Nature 301:279-384.

McKelvey, K. S., K. B. Aubry, and M. K. Schwartz. 2008. Using anecdotal occurrence data for rare or elusive species: the illusion of reality and a call for evidentiary standards. BioScience 58(6):549-555.

Moriarty, K. M. 2014. Habitat Use and Movement Behavior of Pacific Marten (*Martes caurina*) in Response to Forest Management Practices in Lassen National Forest, California. Doctoral Dissertation, Oregon State University, Corvallis Oregon.

Moriarty, K.M., W.J. Zielinski, and E.D. Forsman. 2011. Decline in American Marten Occupancy Rates at Sagehen Experimental Forest, California. The Journal of Wildlife Management 75 (8): 1774-1787.

Moriarty, K. M., W. J. Zielinski, A. G. Gonzales, T. E. Dawson, K. M. Boatner, C. A. Wilson, F. V. Schlexer, K. L. Pilgrim, J. P. Copeland, and M. K. Schwartz. 2009. Wolverine confirmation in California after nearly a century: native or long-distance immigrant? Northwest Science 83(2):154-162.

Moriarty, K. M. 2009. American marten distributions over a 28 year period: relationships with landscape change in Sagehen Creek Experimental Forest, California, USA. M.S. Thesis, Oregon State University, Corvallis, Oregon.

Moritz, C., J. L. Patton, C. J. Conroy, J. L. Parra, G. C. White, and S. R. Beissinger. 2008. Impact of a century of climate change on small-mammal communities in Yosemite National Park, USA. Science 322:261-264.

Mowat, G. 2006. Winter habitat associations of American martens *Martes americana* in interior wet-belt forests. Wildlife Biology 12(1):51-61.

Mulders, R., J. Boulanger, and D. Paetkau. 2007. Estimation of population size for wolverines *Gulo gulo* at Daring Lake, Northwest Territories, using DNA based mark-recapture methods. Wildlife Biology 13(Suppl. 2):38-51.

Perrine, J. D., III. 2005. Ecology of red fox (Vulpes vulpes) in the Lassen Peak region of California, USA. Dissertation. University of California, Berkeley.

Perrine, J. D., J. P. Pollinger, B. N. Sacks, R. H. Barrett, and R. K. Wayne. 2007. Genetic evidence for the persistence of the critically endangered Sierra Nevada red fox in California. Conservation Genetics 8:1083-1095.

Persson, J. A. Landa, R. Andersen, and P. Segerström. 2006. Reproductive characteristics of female wolverines (Gulo gulo) in Scandinavia. Journal of Mammalogy 87(1):75-79.

Potvin, F., L. Belanger and K. Lowell. 2000. Marten habitat selection in a clear-cut boreal landscape. Conservation Biology 14:844-857.

Powell, R. A. 1979. Mustelid spacing patterns: variations on a theme by Mustela Zhurnal Tierpsychologie. 50:153-165.

Powell, R. A. 1994. Structure and spacing in *Martes* populations. Pages 101-121 *in* S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell (editors); Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Powell, R. A., S. W. Buskirk, and W. J. Zielinski. 2003. Fisher and marten (*Martes pennanti* and *Martes americana*). Pp. 635-649 *in* G. A. Feldhamer, B. C. Thompson, and J. A. Chapman, eds. Wild Mammals of North America: Biology, Management, and Conservation, 2nd Edition. The Johns Hopkins University Press, Baltimore, Maryland, and London, England.

Pulliainen, E. 1968. (as cited in White and Barrett 1979). Breeding biology of the wolverine (*Gulo gulo* L.) in Finland. Annales Zoologici Fennici 5(4):338-344.

Robitaille, J.-F., and K. Aubry. 2000. Occurrence and activity of American martens Martes Americana in relation to roads and other routes. Acta Theriologica 45(1):137-143.

Rowland, M. M., M. J. Wisdom, D. H. Johnson, B. C. Wales, J. P. Copeland, and F. B. Edelmann. 2003. Evaluation of landscape models for wolverines in the interior northwest, United States of America. Journal of Mammalogy 84(1):92-105.

Ruggiero, L. F., K. B. Aubry, S. W. Buskirk, L. J. Lyon and W. J. Zielinski, tech. eds. 1994. The Scientific Basis for Conserving Forest Carnivores: American Marten, Fisher, Lynx, and Wolverine in the United States. Gen. Tech. Rep. RM-254. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 184 pp.

Ruggiero, L. F., D. E. Pearson, and S. E. Henry. 1998. Characteristics of American marten den sites in Wyoming. Journal of Wildlife Management 62(2):663-673.

Ruggiero, L. F., K. S. McKelvey, K. B. Aubry, J. P. Copeland, D. H. Pletscher, and M. G. Hornocker. 2007. Wolverine conservation and management. Journal of Wildlife Management 71(7):2145-2146.

Ruth, F. S. 1954. Wolverine seen in Squaw Valley, California. Journal of Mammalogy 35(4):594-595.

Sæther, B-E., S. Engen, J. Persson, H. Brøseth, A. Landa, and T. Willebrand. 2005. Management strategies for the wolverine in Scandinavia. Journal of Wildlife Management 69(3):1001-1014.

Schempf, P. F. and M. White. 1977. Status of six furbearer populations in the mountains of northern California. USDA Forest Service, California Region. 51 pp.

Schwartz, M. K., K. B. Aubry, K. S. McKelvey, K. L. Pilgrim, J. P. Copeland, J. R. Squires, R. M. Inman, S. M. Wisely, and L. F. Ruggiero. 2007. Inferring geographic isolation of wolverines in California using historical DNA. Journal of Wildlife Management 71(7):2170-2179.

Sherburne, S. S. and J. A. Bissonette. 1994. Marten subnivean access point use: response to subnivean prey levels. Journal of Wildlife Management 58:400-405.

Simon, T. L. 1980. An ecological study of the marten in the Tahoe National Forest, California. M.S. Thesis. California State University, Sacramento, California. 140 pp.

Slauson, K. M. 2003. Habitat selection by American martens (Martes Americana) in coastal northwestern California. MS Thesis, Oregon State University, Corvallis, Oregon.

Slauson, K. M., W. J. Zielinski, and J. P. Hayes. 2007. Habitat selection by American martens in coastal California. Journal of Wildlife Management 71(2):458-468.

Spencer, W. D. 1981. Pine marten habitat preferences at Sagehen Creek, California. M.S. Thesis, University of California, Berkeley, California. 121 pp.

Spencer, W. D. and W. J. Zielinski. 1983. Predatory behavior of pine martens. Journal of Mammalogy 64:715-717.

Squires, J. R., J. P. Copeland, T. J. Ulizio, M. K. Schwartz, and L. F. Ruggiero. 2007. Sources and patterns of wolverine mortality in western Montana. Journal of Wildlife Management 71(7):2213-2220.

Soutiere, E. C. 1979. Effects of timber harvesting on marten in Maine. Journal of Wildlife Management 43(4):850-860.

Sumner, L. and J. S. Dixon. 1953. Birds and mammals of the Sierra Nevada. University of California Press, Berkeley, California. 484pp.

USDA Forest Service. Photo Series for Quantifying Natural Forest Residues: Southern Cascades, Northern Sierra Nevada. 1981. 145 pp.

USDA Forest Service. 1993. Ecological Guide to Mixed Conifer Plant Associations. 71 pp.

USDA Forest Service. 1994. Guide to Forested Communities of the Upper Montane in the Central and Southern Sierra Nevada. 164 pp.

Webb, S. M., and M. S. Boyce. 2009. Marten fur harvests and landscape change in west-central Alberta. Journal of Wildlife Management 73(6):894-903.

White, M. and R. H. Barrett. 1979. A review of the wolverine in California with recommendations for management. Final Report, Part A. Prepared for the USDA Forest Service, Region 5: Supplement No. 33, Part A under Master Memorandum of Understanding No. 21-395. Department of Forestry and Resource Management, College of Natural Resources, University of California, Berkley. 71 pp.

Wilbert, C. J., S. W. Buskirk, and K. G. Gerow. 2000. Effects of weather and snow on habitat selection by American martens (*Martes americana*). Canadian Journal of Zoology 78:1691-1696.

Wildlife Management Institute. 1974. (*as cited* in Shempf and White 1977) Wolverine expands range. Wildlife Management Institute, Washington, D.C. Outdoor News Bulletin 28(9):4.

Yocum, C. F. 1973. Wolverine records in the Pacific coastal states and new records for northern California. California Fish and Game 59:207-209.

Zeiner, D. C., W. F. Laudenslayer, Jr., K. E. Mayer and M. White. 1990. California Wildlife: Volume III: Mammals. California Dept. of Fish and Game, Sacramento CA. 407 pp.

Zielinski, W. J., W. D. Spencer and R. H. Barrett. 1983. Relationship between food habits and activity patterns of pine martens. Journal of Mammalogy 64:387-396.

Zielinski, W. J., T. E. Kucera, and R. H. Barrett. 1995. Current distribution of fishers, *Martes pennanti*, in California. California Fish and Game 81, 104-112.

Zielinski, W. J., and R. T. Golightly. 1996. The status of marten in redwoods: is the Humboldt marten extinct? Pp. 115-119 in J. LeBlanc, ed. Conference on Coast Redwood Forest Ecology and Management. Humboldt State University, Arcata, California.

Zielinski, W. J., R. H. Barrett, and R. L. Truex. 1997. Southern Sierra Nevada fishers and marten study: Progress Report IV. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, California. 37 pp.

Zielinski, W. J., F. V. Schlexer, R. L. Truex and L. A. Campbell. 1999. A preliminary assessment of live-trapping as a technique to estimate the proportion of reproductive female fishers. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Arcata, California. 18 pp.

Zielinski, W. J., K. M. Slauson, C. R. Carroll, C. J. Kent, and D. G. Kudrna. 2001. Status of American martens in coastal forests of the Pacific states. Journal of Mammalogy 82, 478-490.

Zielinski, W. J., and N. P. Duncan. 2004. Diets of sympatric populations of American martens (*Martes americana*) and fishers (*Martes pennanti*) in California. Journal of Mammalogy 85(3):470-477.

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. Journal of Biogeography 32:1385-1407.

Zielinski, W. J., K. M. Slauson, and A. E. Bowles. 2008. Effects of off-highway vehicle use on the American marten. Journal of Wildlife Management 72(7):1558-1571.

Zielinski, W.J. 2013. The Forest Carnivores: Fisher and Marten. Chapter 7.1 in: Long, J., C. Skinner, M. North, P. Winter, W. Zielinski, C. Hunsaker, B. Collins, J. Keane, F. Lake, J. Wright, E. Moghaddas, A. Jardine, K. Hubbert, K. Pope, A. Bytnerowicz, M. Fenn, M. Busse, S. Charnley, T. Patterson, L. Quinn-Davidson and H. Safford. 2013. Science Synthesis to Support Forest Plan Revision in the Sierra Nevada and Southern Cascades. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 504 p.

Pallid Bat

Baker, M., M. Lacki, G. Falxa, P. Droppelman, R. Slack, and S. Slankard. Habitat use of pallid bats in coniferous forests of northern California. Northwest Science 82(4).

Pallid Bat Recovery Team. 2008. Recovery Strategy for the Pallid Bat (*Antrozous pallidus*) in British Columbia. Prepared for the B.C. Ministry of Environment, Victoria B.C. 18 pp.

Barbour, R.W. and W.H. Davis. 1969. Bats of America. University Press of Kentucky. 283 pp.

Brown, P. 1996. Presentation at the Natural History and Management of Bats in California and Nevada Conference. The Wildlife Society Western Section, Nov. 13-15, 1996. Sacramento, California.

Burt, W., and R. Grossenheider. 1980. Mammals, third edition. Peterson Field Guide. Houghton-Mifflin Company. Boston, Massachusetts.

CNDDB. California Department of Fish and Game, Biogeographic Data Branch. 2011. California Natural Diversity Database. Sacramento, CA. Data downloaded November 2011.

CWHR. California Department of Fish and Game, California Interagency Wildlife Task Group. 2008. CWHR version 8.2 personal computer program. Sacramento, CA.

Chapman, K., K. McGuiness, and R. Brigham. 1994. The status of the pallid bat in British Columbia. Wildlife Branch, Ministry of Environment, Land, and Parks. Victoria, BC. 35 pages.

Ferguson, H., and J. Azerrad. 2004. Pallid bat *In* Management Recommendations for Washington's Priority Species – Volume V: Mammals. Washington Department of Fish and Wildlife. Olympia, WA.

Hermanson, J., and T. O'Shea. 1983. *Antrozous pallidus*. Mammalian Species. No. 213.Pages 1-8. Published by the American Society of Mammalogists.

https://www.whitenosesyndrome.org/resources/map

Johnston, D., and J. Gworek. 2006. [ABS]. Pallid bat (*Antrozous pallidus*) habitat use in a coniferous forest in northeastern California. Bat Research News 47:4,.

Miner, K., and D. Stokes. 2005. Bats in the south coast ecoregion: status, conservation issues, and research needs. *In*: Planning for Biodiversity: Bringing Research and Management

Together. Kus, B., and J. Beyers, technical coordinators. Gen. Tech. Rep. PSW-GTR-195. US Forest Service, Pacific Southwest Research Station. Albany, CA. 274 pages.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: December 22, 2011).

Philpott, W. 1997. Summaries of the life histories of California bat species. Sierra National Forest, Pineridge Ranger District. Unpublished doc.

Pierson, E.D. 1996. Presentation to the Natural History and Management of Bats in California and Nevada Workshop. November 13-15. Sacramento, CA.

Pierson, E., M. Wackenhut, J. Altenbach, P. Bradley, P. Call, D. Genter, C.. Harris, B. Keller, B. Lengus, L. Lewis, B. Luce, K. Navo, J. Perkins, S. Smith, and L. Welch. 1999. Species conservation assessment and strategy for Townsend's big-eared bat (*Corynorhinus townsendii townsendii* and *Corynorhinus townsendii pallescens*). Idaho Conservation Effort, Idaho Department of Fish and Game, Boise, ID.

Rambaldini, D. 2005. Species account for the pallid bat (2005 update on the 1998 account by R. Sherwin). Western Bat Working Group.

Stephenson, J., and G. Calcarone. 1999. Southern California mountains and foothills assessment: habitat and species conservation issues. Gen. Tech. Rep. GTR-PSW-175. US Forest Service, Pacific Southwest Research Station. Albany, CA. 402 pages.

USFS. 2010. Deputy Chief's letter to Regional Foresters, Station Directors, Area Director, IITF Director, Deputy Chiefs and WO Directors regarding implementation a Forest Service white-nose syndrome interim response strategy. File Code 2600. July 28, 2010. US Forest Service, Washington Office. Washington, D.C.

U.S. Department of Agriculture Forest Service. (USDA FS). 2011. Natural Resource Information System (NRIS) database. Accessed November 2011.

USFWS. 2011. A National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-Nose Syndrome in Bats. US Fish and Wildlife Service, Northeast Region. Hadley, MA. 21 pages.

Vaughan, T., and T. O'Shea. 1976 Roosting ecology of the pallid bat, *Antrozous pallidus*. Journal of Mammalogy 57(1).

Zeiner, David C., William F. Laudenslayer, Jr., Kenneth E. Mayer, and Marshall White, eds. 1990. California's Wildlife. Volume III. Mammals. California Statewide Wildlife Habitat Relationship System. Department of Fish and Game, The Resources Agency, Sacramento, California. 407 pages.

Townsend's Big-eared Bat

Arnett, E., W. Brown, W. Erickson, J. Fiedler, B. Hamilton, T. Henry, A. Jain, G. Johnson, J. Kerns, R. Koford, C. Nicholson, T. O'Connell, M. Piorkowski, and R. Tankersley, Jr.

2008. Patterns of bat fatalities at wind energy facilities in North America. Journal of Wildlife Management, 72(1):61-78.

Barbour, R.W. and W.H. Davis. 1969. Bats of America. University Press of Kentucky. 283 pp.

Bogan, M. 2000. Western bats and mining. *In:* K. Vories and D. Throgmorton, eds., Proceedings of bat conservation and mining: a technical interactive forum. U.S.D.I Office of Surface Mining, and Coal Research Center at Southern Illinois University. 374 pages.

Bradley, P., M. O'Farrell, J. Williams, and J. Newmark. 2006. Eds. The Revised Nevada Bat Conservation Plan. Nevada Bat Working Group. Reno, Nevada. 216 pages.

Burt, W., and R. Grossenheider. 1980. Mammals, third edition. Peterson Field Guide. Houghton-Mifflin Company. Boston, Massachusetts.

CNDDB. California Department of Fish and Game, Biogeographic Data Branch. 2011. California Natural Diversity Database. Sacramento, CA. Data downloaded November 2011.

California Department of Fish and Game, California Interagency Wildlife Task Group. 2008. CWHR version 8.2 personal computer program. Sacramento, CA.

Ellison, L., T. O'Shea, M. Bogan, A. Everette, and D. Schneider. 2003a. Existing data on colonies of bats in the United States: Summary and Analysis of the U.S. Geological Survey's Bat Population Database. *In*: O'Shea, T.J. and Bogan, M.A., eds., Monitoring trends in bat populations of the United States and territories: problems and prospects: U.S. Geological Survey, Biological Resources Discipline, Information and Technology Report, USGS/BRD/ITR--2003–0003, 274 p.

Ellison, L., M. Wunder, C. Jones, C. Mosch, K. Navo, K. Peckham, J. Burghardt, J. Annear, R. West, J. Siemers, R. Adams, and E. Brekke. 2003b. Colorado bat conservation plan. Colorado Committee of the Western Bat Working Group. Available at: http://www.wbwg.org/conservation/conservationplans.html.

Fellers, G. and E. Pierson. 2002. Habitat use and foraging behavior of Townsend's big-eared bat (*Corynorhinus townsendii*) in coastal California. Journal of Mammalogy 83(1).

Geluso, K. 1978. Urine concentrating ability and renal structure of insectivorous bats. Journal of Mammalogy 59(2).

Gruver, J.C. and D.A. Keinath. 2006. Townsend's Big-eared Bat (*Corynorhinus townsendii*): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/townsendsbigearedbat.pdf.

Jameson, E.W. Jr. and Hans J. Peeters. 1988. California Mammals. University of California Press. Berkeley, California.

Kunz, T.H. and R.A. Martin. 1990. *Plecotus townsendii*. Mammalian Species. No. 175. pp 1-6. Published by the American Society of Mammalogists.

Kunz, T., E. Arnett, W. Erickson, A. Hoar, G. Johnson, R. Larkin, M. Strickland, R. Thresher, and M. Tuttle. 2007. Ecological impacts of wind energy development on bats: questions, research needs, and hypotheses. Frontiers in Ecology and the Environment 5: 315–324. http://dx.doi.org/10.1890/1540-9295(2007)5[315:EIOWED]2.0.CO;2

Miner, K., and D. Stokes. 2005. Bats in the south coast ecoregion: status, conservation issues, and research needs. *In*: Planning for Biodiversity: Bringing Research and Management Together. Kus, B., and J. Beyers, technical coordinators. Gen. Tech. Rep. PSW-GTR-195. US Forest Service, Pacific Southwest Research Station. Albany, CA. 274 pages.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: December 22, 2011).

Philpott, W. 1997. Summaries of the life histories of California bat species. Sierra National Forest, Pineridge Ranger District. Unpublished doc.

Piaggio, A. 2005. Species account for the Townsend's big-eared bat (2005 update on the 1998 account by R. Sherwin). Western Bat Working Group.

Pierson, E.D. 1996. Presentation to the Natural History and Management of Bats in California and Nevada Workshop. November 13-15. Sacramento, CA.

Pierson, E.D., and W.E. Rainey. 1998. Distribution, status, and management of Townsend's big-eared bat (*Corynorhinus townsendii*) in California. California Department of Fish and Game, The Resources Agency, Sacramento, California.

Pierson, E.D., W.E. Rainey, and D.M. Koontz. 1991. Experimental mitigation for Townsend's big-eared bat at the McLaughlin Mine in California. From Proceedings V: Issues and technology in the management of impacted wildlife, April 8-10, 1991. Snowmass, Colorado. Thorne Ecological Institute, Boulder, Colorado.

Pierson, E.D., and W.E. Rainey. 2007. Bat distribution in the forested region of northwestern California. California Department of Fish and Game, Wildlife Management Division, Nongame Bird and Mammal Section. Sacramento, California. 46 pages.

Pierson, E., M. Wackenhut, J. Altenbach, P. Bradley, P. Call, D. Genter, C.. Harris, B. Keller, B. Lengus, L. Lewis, B. Luce, K. Navo, J. Perkins, S. Smith, and L. Welch. 1999. Species conservation assessment and strategy for Townsend's big-eared bat (*Corynorhinus townsendii townsendii* and *Corynorhinus townsendii pallescens*). Idaho Conservation Effort, Idaho Department of Fish and Game, Boise, ID.

Sherwin, R., W. Gannon, J. Altenbach, and D. Stricklan. 200. Roost fidelity of Townsend's big-eared bats in Utah and Nevada . Transactions of the Western Section of the Wildlife Society 36:15-20.

Sherwin, R., W. Gannon, and J. Altenbach. 2003. Managing complex systems simply: understanding inherent variation in the use of roosts by Townsend's big-eared bat. Wildlife Society Bulletin (31(1).

Siedman, V.M., and C.J. Zabel. 2001. Bat activity along intermittent streams in northwestern California. Journal of Mammalogy 82(3).

Thomas, D., and S. West,. 1989. Sampling methods for bats. *In*: Ruggiero, Leonard F.; Carey, Andrew B., tech. eds.; Wildlife-habitat relationships: sampling procedures for Pacific Northwest vertebrates Gen. Tech. Rep. PNW-GTR-243. US Forest Service, Pacific Northwest Research Station. Portland, Oregon. 20 pages.

U.S. Department of Agriculture Forest Service. 2010. Deputy Chief's letter to Regional Foresters, Station Directors, Area Director, IITF Director, Deputy Chiefs and WO Directors regarding implementation a Forest Service white-nose syndrome interim response strategy. File Code 2600. July 28, 2010. US Forest Service, Washington Office. Washington, D.C.

U.S. Department of Agriculture Forest Service. (USDA FS). 2011. Natural Resource Information System (NRIS) database. Accessed November 2011.

U.S. Department of Interior, Fish and Wildlife Service (USFWS). 2011. A National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-Nose Syndrome in Bats. US Fish and Wildlife Service, Northeast Region. Hadley, MA. 21 pages.

Zeiner, David C., William F. Laudenslayer, Jr., Kenneth E. Mayer, and Marshall White, eds. 1990. California's Wildlife. Volume III. Mammals. California Statewide Wildlife Habitat Relationship System. Department of Fish and Game, The Resources Agency, Sacramento, California. 407 pages.

Fringed Myotis

Altenbach, J.S., and E.D. Pierson. 1995. The importance of mines to bats: an overview. pp. 7-18, *in* B.R. Riddle, ed. Inactive mines as bat habitat: guidelines for research, survey, monitoring and mine management in Nevada, Biological Resources Research Center, University of Nevada, Reno.

Angerer, L. M. and E. D. Pierson. *Draft*. Fringed myotis (*Myotis thysanodes*) species account. 13pp.

Baker, J.K. 1962. Notes on the Myotis of the Carlsbad Caverns. Journal of Mammalogy, 43:427-428.

Barbour, R.W. and W.H. Davis. 1969. Bats of America. University of Kentucky Press, Lexington, Ky, 286 pp.

Black, H.L. 1974. A north temperate bat community: structure and prey populations. Journal of Mammalogy, 55:138-157.

Brigham, R.M. 1991. Flexibility in foraging and roosting behavior by the big brown bat (*Eptesicus fuscus*). Canadian Journal of Zoology, 69:117-121.

Cahalane, V.H. 1939. Mammals of the Chiricahua Mountains, Cochise County, Arizona. Journal of Mammalogy, 20:418-440.

Chung-MacCoubrey, A.L. 1996. Bat species composition and roost use in pinyon-juniper woodlands of New Mexico. Pp. 118-123, in R.M.R. Barclay and M.R. Brigham, Editors.

Bats and Forests Symposium, October 19-21, 1995, Victoria, British Columbia, Canada, Research Branch, Ministry of Forests, Victoria, British Columbia, Working Paper 23/1996.

Clark, D. R. 1981. Death in bats from DDE, DDT or dieldrin: diagnosis via residues in carcass fat. Bulletin of environmental contamination and toxicology, 26(1): 367-374.

Clark Jr, D. R., LaVal, R. K., & Swineford, D. M. 1978. Dieldrin-induced mortality in an endangered species, the gray bat (*Myotis grisescens*). Science, 199(4335): 1357-1359.

Clark, D. R., Bunck, C. M., Cromartie, E., & Laval, R. K. 1983. Year and age effects on residues of dieldrin and heptachlor in dead gray bats, Franklin County, Missouri—1976, 1977, and 1978. Environmental Toxicology and Chemistry, 2(4): 387-393.

Cockrum, E.L., and B.F. Musgrove. 1964. Additional records of the Mexican big-eared bat, P*lecotus phyllotis* (Allen), from Arizona. Journal of Mammalogy, 45:472-474.

Cockrum, E.L., and E. Ordway. 1959. Bats of the Chiricahua Mountains, Cochise County, Arizona. Amer. Mus. Novitates, 1938:1-35

CWHR. California Department of Fish and Game, California Interagency Wildlife Task Group. 2008. CWHR version 8.2 personal computer program. Sacramento, CA.

Easterla, D.A. 1966. Yuma myotis and fringed myotis in southern Utah. Journal of Mammalogy, 47:350-351.

Easterla, D.A. 1973. Ecology of the 18 species of Chiroptera at Big Bend National Park, Texas. Northwest Missouri State Univ. Studies, 34:1-165.

Hirshfeld, J.R., and M.J. O'Farrell. 1976. Comparisons of differential warming rates and tissue temperatures in some species of desert bats. Comp. Biochem. Physiol., 55A:83-87.

Jones, C. 1965. Ecological distribution and activity periods of bats of the Mogollon Mountains area of New Mexico and adjacent Arizona. Tulane Studies Zool., 12:93-100.

Keinath, D.A. 2004. Fringed myotis (*Myotis thysanodes*): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/fringedmyotis.pdf.

Kunz, T.H. 1982. Roosting ecology of bats *in* T.H. Kunz, editor, Ecology of bats. Plenum Press, New York, USA. Pages 1-55.

Lewis, S.E. 1995. Roost fidelity of bats: a review. Journal of Mammalogy, 76:481-496.

Miner, K. and P. Brown. 1996. A report on the southern California forest bat survey and radio-telemetry study of 1996. Contract Report for USDA Forst Service, San Diego, CA, 13 pp.

Miner, K., P. Brown, R. Berry, C. Brown-Buescher, A. Kisner, S. Remington, D. Simons, D.

Stokes, J. Stephenson, and L. Underwood. 1996. Habitat use by *Myotis evotis* and *M. thysanodes* in a southern California pine-oak woodland. Bat Research News, 37(4):141.

NatureServe. 2012. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe. Arlington, Virginia. Available http://www.natureserve.org/explorer. (Accessed: October 17, 2012)

O'Farrell, M.J., and E.H. Studier. 1973. Reproduction, growth, and development in *Myotis thysanodes* and *M. lucifugus* (Chiroptera: Vespertilionidae). Ecology, 54(1):18-30.

O'Farrell, M.J., and E.H. Studier. 1975. Population structure and emergence activity patterns in *Myotis thysanodes* and *M. lucifugus* (Chiroptera: Vespertilionidae) in northeastern New Mexico. Amer. Midland Nat., 93:368-376.

O'Farrell, M.J. and E.H. Studier. 1980. *Myotis thysanodes*. American Society of Mammalogists, Mammalian Species, 137:1-5.

Pierson, E.D., and P.A. Heady. 1996. Bat surveys of Giant Forest Village and vicinity, Sequoia National Park. Report for National Park Service, Denver Service Center, Denver, CO, 27 pp.

Pierson, E., M. Wackenhut, J. Altenbach, P. Bradley, P. Call, D. Genter, C.. Harris, B. Keller, B. Lengus, L. Lewis, B. Luce, K. Navo, J. Perkins, S. Smith, and L. Welch. 1999. Species conservation assessment and strategy for Townsend's big-eared bat (Corynorhinus townsendii townsendii and Corynorhinus townsendii pallescens). Idaho Conservation Effort, Idaho Department of Fish and Game, Boise, ID.

Pierson, E.D., W.E. Rainey, and C.J. Corben. 2001. Seasonal patterns of bat distribution along an altitudinal gradient in the Sierra Nevada. Report to California State University at Sacramento Foundation, Yosemite Association, and Yosemite Fund, 70 pp.

Pierson, E. D., W. E. Rainey, and L.S. Chow. 2006. Bat use of the giant sequoia groves in Yosemite National Park. Report to Yosemite Fund, San Francisco, CA and Yosemite National Park, El Portal, CA, 154 pp.

Rabe, M.J., T.E. Morrell, H. Green, J.C. DeVos, Jr., and C.R. Miller. 1998. Characteristics of ponderosa pine snag roosts used by reproductive bats in northern Arizona. Journal of Wildlife Management, 62:612-621.

Rainey, W.E. and E.D. Pierson. 1996. Cantara spill effects on bat populations of the upper Sacramento River, 1991-1995. Report to California Department of Fish and Game, Redding, CA, (Contract # FG2099R1). 98 pp.

Roest, A.I. 1951. Mammals of the Oregon Caves area, Josephine County. Journal of Mammalogy, 32:345-351.

Studier, E.H., and M.J. O'Farrell. 1972. Biology of *Myotis thysanodes* and *M. lucifugus* (Chiroptera: Vespertilionidae) – I. Thermoregulation. Comp. Biochem. Physiol., 41A:567-595.

U.S. Department of Agriculture Forest Service (USDA FS). 2005. Fringed myotis. In: Species Accounts – Animals. Available at: http://www.fs.fed.us/r5/scfpr/projects/lmp/docs/species-animals.pdf.

USFS. 2010. Deputy Chief's letter to Regional Foresters, Station Directors, Area Director, IITF Director, Deputy Chiefs and WO Directors regarding implementation a Forest Service white-nose syndrome interim response strategy. File Code 2600. July 28, 2010. US Forest Service, Washington Office. Washington, D.C.

U.S. Department of Agriculture Forest Service (USDA FS). 2011. Natural Resource Information System (NRIS) database.

USFWS. 2011. A National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-Nose Syndrome in Bats. US Fish and Wildlife Service, Northeast Region. Hadley, MA. 21 pages.

Weller, T.J., and C.J. Zabel. 2001. Characteristics of fringed *Myotis* day roosts in northern California. Journal of Wildlife Management, 65:489-497.

Weller, T. 2005. Species account for the fringed myotis (update of the 1998 account by P. Bradley and M. Potts). Western Bat Working Group.

Wilson, B. W., Hooper, M. J., Littrell, E. E., Detrich, P. J., Hansen, M. E., Weisskopf, C. P., & Seiber, J. N. 1991. Orchard dormant sprays and exposure of red-tailed hawks to organophosphates. Bulletin of environmental contamination and toxicology, 47(5): 717-724.

California Red-legged Frog

Barry, S. J. 1999. A study of the California Red-legged frog (Rana draytonii) of Butte County, California. PAR Environmental Services, Technical Report No. 3. Sacramento, California.

Barry, S. J. and G. M. Fellers. 2013. History and status of the California red-legged frog (*Rana draytonii*) in the Sierra Nevada, California, USA. Herpetological Conservation and Biology 8:456-502.

Briggs, C.J., V.T. Vredenburg, R.A. Knapp, and L.J. Rachowicz. 2005. Investigating the population-level effects of chytridiomycosis: an emerging infectious disease of amphibians. Ecology 86:3149–3159.

Calef, G. W. 1973. Natural mortality of tadpoles in a population of *Rana aurora*. Ecology 54: 741-758. California Academy of Sciences. 1992. Copy of locality records from the data base of California Academy of Sciences.

Fellers, G. M. 1997. 1997 Aquatic Amphibian Surveys—Tahoe National Forest. Biological Resources Division, USGS, Point Reyes National Seashore, Point Reyes, CA. Unpublished Report Available from: Tahoe National Forest, Nevada City, CA.

Fellers, G. M.; R. A. Cole, D. M. Reinitz, and P. M. Kleeman. 2011. Amphibian chytrid fungus (Batrachochytrium dendrobatidis) in coastal and montane California, USA Anurans. Herpetological Conservation and Biology 6:383-394.

Fellers, G. M. and K. L. Freel. 1995. A Standardized Protocol for Surveying Aquatic Amphibians. Technial Report NPS/WRUC/NRTR-95-001. National Biological Service, Cooperative Park Studies Unit, University of California, Davis, CA. 23pp.

Fellers, G.M., D.E Green, and J.E. Longcore. 2001. Oral chytridiomycosis in Mountain Yellow-legged Frogs (*Rana muscosa*). Copeia 2001:945–953.

Fellers, G. M. and P. M. Kleeman. 2006. Diurnal versus nocturnal surveys for California red-legged frogs. Journal of Wildlife Management 70:1805-1808.

Fellers, G., and P. Kleeman. 2007. California Red-legged Frog (*Rana draytonii*) movement and habitat use: Implications for conservation. Journal of Herpetology 41:271-281.

Freel. K. 1997. Herpetologist with USGS, Point Reyes, CA [personal communication with Tahoe National Forest Fisheries Biologist].

Hayes, M. P. and M. R. Jennings. 1986. Decline of ranid frog species in western North America: are bullfrogs (*Rana catesbeiana*) responsible? Journal of Herpetology 20(4):490-509.

Hayes, M. P. and M. R. Jennings. 1988. Habitat correlates of distribution of the California red-legged frog (*Rana aurora draytonii*) and the foothillyellow-legged frog (*Rana boylii*): implications for management. Proceedings of the Symposium on Management of Amphibians, Reptiles and Small Mammals in North America. Gen. Tech. Rpt. RM-166, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado. pp 144-158.

Jennings, M. K. 1988. Natural history and decline of native ranids in California. Proceedings of the Conference on California Herpetology. Southwestern Herpetologists Society. p. 61-72.

Jennings, M. R. 1997. Research Associate with California Academy of Sciences, San Francisco, CA [personal communication with Tahoe National Forest Biologist].

Jennings, M. R. and M. P. Hayes. 1985. Pre-1900 overharvest of California red-legged frogs (*Rana aurora draytonii*) : the inducement for bullfrog (*Rana catesbeiana)* introduction. Herpetologica 41 (1): 94-103.

Jennings, M. R. and M. P. Hayes. 1992. Copy of locality records from the data base of Mark R. Jennings and Marc P. Hayes for use in support of the petition to list *Rana aurora draytonii* under the endangered Species Act. 30pp.

Jennings, M. R. and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. California Department of Fish and Game, Final Report. 255 pp.

Licht, L. E. 1971. Breeding habits and embryonic thermal requirements of the frog, *Rana aurora aurora* and *Rana pretiosa pretiosa,* in the Pacific Northwest. Ecology 52(1):116-124.

Martin, D. L. 1992. Sierra Nevada Anuran Guide. Canorus Ltd. Ecological Research Team. Canorus Ltd. Press, San Jose, California. 28 pp.

McMahon, T. A.; L. A. Brannelly, M. W. H. Chatfield, P. T. J. Johnson, M. B. Joseph, V. J.

McKenzie, C. L. Richards-Zawacki, M. D. Venesky, and J. R. Rohr. 2013. Chytrid fungus Batrachocytrium dendrobatidis has nonamphibian hosts and releases chemicals that cause pathology in the absence of infection. Proceedings of the National Academy of Sciences of the United States of America 110:210. Available at: www.pnas.org/cgi/doi/10.1073/pnas.1200592110.

Moyle, P. B. 1973. Effects of introduced bullfrogs, *Rana catesbeiana*, on the native frogs of the San Joaquin Valley, California. Copeia: 18-22.

Padgett-Flohr, G.E. 2008. Pathogenicity of Batrachochytrium dendrobatidis in two threatened California amphibians: Rana draytonii and Ambystoma californiense. Herpetological Conservation and Biology 3(2):182-191.

Padgett-Flohr, G.E., and R.L. Hopkins, II. 2009. *Batrachochytrium dendrobatidis*, a novel pathogen approaching endemism in central California. Diseases of Aquatic Organisms 83:1–9.

Padgett-Flohr, G.E., and R.L. Hopkins, II. 2010. Landscape epidemiology of *Batrachochytrium dendrobatidis* in central California. Ecography 33:688–697.

Rathbun, G. B., M. R. Jennings, T. G. Murphey and N. R. Siepel. 1993. Status and ecology of sensitive aquatic vertebrates in lower San Simeon and Pico Creeks, San Luis Obispo Co., Calif. USFWS National Ecology Research Center., San Simeon, CA. Prepared for California Department of Parks and Recreation. 103 pp.

Storer, T. I. 1925. A synopsis of the Amphibia of California. Univ. Calif. Publ. Zool. 27: 1-342.

Tatarian, P. J. 2008. Movement patterns of California red-legged frogs (*Rana draytonii*) in an inland California environment. Herpetological Conservation and Biology 3:155-169.

Tatarian. 2008. California red-legged frog telemetry study; Hughes Pond, Plumas National Forest. Annual Report, Option Year 3 to: U. S. Fish and Wildlife Service, 2800 Cottage Way, Sacrament, CA and U.S. Forest Service, Plumas National Forest, 875 Mitchell Avenue, Oroville, CA. Prepared September 28, 2008 by: Trish and Greg Tatarian, Wildlife Research Associates, 1119 Burbank Avenue, Santa Rosa, CA 95407.

Tatarian, P. and G. Tatarian. 2010. Chytrid infection of *Rana draytonii* in the Sierra Nevada, California, USA. Herpetological Review 41: 325-327.

USDI Fish and Wildlife Service. 1996. Endangered and Threatened Wildlife and Plants; Determination of Threatened Status for the California Red-Legged Frog. Federal Register 61(101):25813-25833.

USDI Fish and Wildlife Service. 2000. Endangered and Threatened Wildlife and Plants; Proposed Designation of Critical Habitat for the California Red-Legged Frog (*Rana aurora draytonii*. Federal Register 65(176):54892-54932.

USDI Fish and Wildlife Service. 2002. Recovery Plan for the California Red-legged Frog (*Rana aurora draytonii*). U. S. Fish and Wildlife Service, Portland, OR. 173 pp.

USDI Fish and Wildlife Service. 2010. Endangered and Threatened Wildlife and Plants; Revised Designation of Critical Habitat for California Red-Legged Frog; Final Rule 75(51):12816-12959.

Vindum, Tierney, and Koo. 1997. Amphibian and reptile surveys in the Tahoe National Forest: The result of CCSA-05-97-17-024. Unpublished Report Available at: Tahoe National Forest, Nevada City, CA.

Vindum, J. V. and M. S. Koo. 1998. Amphibian and reptile surveys in the Plumas and Tahoe National Forests: The result of CCSA-05-98-17-123. California Academy of Sciences, CA. Unpublished Report Available at: Tahoe National Forest, Nevada City, CA.

Vindum, J. F. and M. S. Koo. 1999. Amphibians and reptiles of the Tahoe National Forest: historical collections and results of 1997-1999 California Academy of Sciences surveys. Unpublished Report Available at: Tahoe National Forest, Nevada City, CA.

Young, M. 2015. Personal Communications. Natural Resource Restoration Ecologist. Westervelt Ecological Services, Western Region. Sacramento, California.

Lahontan Cutthroat Trout

Behnke, R. J. 1979. Monograph of the native trouts of the genus *Salmo* of western North America. U.S. Department of Agriculture, Forest Service, Lakewood, Colorado. 2155 pp.

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society, Bethesda, Maryland. 275 p.

Calhoun, A. J. 1942. The biology of the black-spotted trout (*Salmo clarki henshawi*) (Gill and Jordan) in two Sierra Nevada lakes. Dissertation. Stanford University, Palo Alto, California. 218 pp.

Coffin, P. D. 1981. Distribution and life history of the Lahontan/Humboldt cutthroat trout, Humboldt River drainage basin. Nevada Department of Wildlife, Reno, Nevada. 69 pp.

Coffin, P. D. 1983. Lahontan cutthroat trout fishery management plan for the Humboldt River Basin. Federal Aid to Fish Restoration Project Report F-20-17, Study 1X, Job No. 1-P-1. Nevada Department of Wildlife, Reno, Nevada. 33 pp.

Coffin, P. D. 1988. Nevada's native salmonid program status, distribution and management. Nevada Department of Wildlife, Reno, Nevada. 400 pp.

Coleman, M. E., and V. K. Johnson. 1988. Summary of trout management at Pyramid Lake, Nevada, with emphasis on Lahontan cutthroat trout, 1954-1987. American Fisheries Society Symposium 4:107-115.

Cowan, W. 1983. Annual fisheries management report FY-83' Summit Lake Indian Reservation, Humboldt Country, Nevada. Unpublished report. Summit Lake Paiute Tribe, Winnemucca, Nevada. 37 pp.

French, J., and J. Curran. 1991. DRAFT Lahontan cutthroat trout fishery management plan for the Quinn River drainage basin. Nevada Department of Wildlife, Reno, Nevada. 46 pp.

Gerstung, E. R. 1986. Fishery management plant for Lahontan cutthroat trout (*Salmo clarki henshawi*) in California and western Nevada water. Inland Fisheries Administrative Report No. 86-, Federal Aid Project F33-R-11, The Resources Agency, California Department of Fish and Game. 54 pp.

Gerstung, E. R. 1988. Status, life history, and management of the Lahontan cutthroat trout. American Fisheries Society Symposium 4:93-106.

Hickman, T., and R. F. Raleigh. 1982. Habitat suitability index models: cutthroat trout. Report number USFWS/OBS-82/10.5. USDI, Fish and Wildlife Service, Western Energy and Land Use Team, Office of Biological Services, Washington, D.C.

Johnson, G., D. Bennett, and T. Bjornn. 1983. Juvenile emigration of Lahontan cutthroat trout in the Truckee River/Pyramid Lake system. Pages 90-175. In: Restoration of a Reproductive Population of Lahontan Cutthroat Trout (*Salmo clarki henshawi*) to the Truckee River/Pyramid Lake System. Special Report U.S. Department of the Interior, Fish and Wildlife Service, Great Basin Complex Office, Reno, Nevada.

Koch, D. L., and J. J. Cooper, E. L. Lider, R. L. Jacobson and R. J. Spencer. 1979. Investigations of Walker Lake, Nevada: Dynamic ecological relationships. Desert Research Center, University of Nevada, Reno. 191 pp.

Lea, T. N. 1968. Ecology of the Lahontan cutthroat trout, *Salmo clarki henshawi*, in Independence Lake, California. Master's thesis. University of California, Berkeley. 95 pp.

McAfee, W. R. 1966. Lahontan cutthroat trout. Pages 225-231 in A. Calhoun, editor. Inland Fisheries management. California Department of Fish and Game, Sacramento, California.

Moyle, P. B. 1976. Inland fishes of California. University of California Press, Berkeley, California. 405 pages.

Moyle, P.B., J.V.E. Katz, and R.M. Quiñones. 2011. Rapid decline of California's native inland fishes: a status assessment. Biological Conservation 144:2414–2423.

Rankel, G. L. 1976. Fishery management program, Summit Lake Indian Reservation, Humboldt County, Nevada. Special Report of the U.S. Fish and Wildlife Service, Division of Fishery Services, Reno, Nevada. 35 pp.

Ray, C., M.M. Peacock, and J.B. Dunham. 2007. Demographic and population dynamics of Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*) stream populations in eastern Nevada. Final Report to the U.S. Fish and Wildlife Service, Reno, Nevada. Cooperative Agreement FSW 14–48–0001–95646. 205 pp.

Rissler, P.H., G.G. Scoppettone, and S. Shea. 2006. Life history, ecology, and population viability analysis of the Independence Lake strain Lahontan cutthroat trout (*Oncorhynchus clarkii henshawi*). Final Report. U.S. Geological Survey, Western Fisheries Research Center, Reno, Nevada. 68 pp.

Rissler, Pete, USGS. Personal communication. April, 2014.

Roberts, B. C., and R. G. White. 1992. Effects of angler wading on survival of trout eggs and pre-emergent fry. North American Journal of Fisheries Management 12:450-459.

Scoppettone, G.G., P. Rissler, S. Shea, and W. Somer. 2012. Effects of brook trout removal from a spawning stream on an adfluvial population of Lahontan cutthroat trout. North American Journal of Fisheries Management 32:586–596.

Sigler, W. F., W. T. Helm, P. A. Kucera, S. Vigg, and G. W. Workman. 1983. Life history of the Lahontan cutthroat trout, *Salmo clarki henshawi*, in Pyramid Lake, Nevada. Great Basin Naturalist 43:1-29.

Sigler, W. F., and J. W. Sigler. 1987. Fishes of the Great Basin: A Natural History. University of Nevada Press, Reno, Nevada. 425 pp.

Somer, W. 1998. Lahontan Cutthroat Trout, A Threatened Resource. Department of Fish and Game Wild Trout Project winter 1998. Vol. 3, No. 2.

Sumner, F. H. 1940. The decline of the Pyramid Lake fishery. Transactions of the American Fisheries Society 69:216-224.

USDA Forest Service. 1990. Tahoe National Forest Land and Resources Management Plan.

USDI Fish and Wildlife Service (USFWS). 1995. Recovery plan for the Lahontan cutthroat trout. Portland, Oregon. 108 pp.

USDI Fish and Wildlife Service (USFWS). 2003. Short-term action plan for Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*) in the Truckee River basin. U.S. Fish and Wildlife Service, Reno, Nevada. August, 2003. i–iv + 71 pp.

USDI Fish and Wildlife Service (USFWS). 2009. 5-Year Review: Summary and Evaluation. Lahontan cutthroat trout (*Oncorhynchus clarkii henshawi*). Region 8, Sacramento, California. March 30, 2009. 199 pp.

Western Pond Turtle

Buskirk, J. 2002. The western pond turtle, *Emys marmorata*. Radiata 11(3): 3-30.

Crump, D.E. 2001. Western pond turtle (*Clemmys marmorata pallida*) nesting behavior and habitat use. Master's Thesis. Paper 2210. http://scholarworks.sjsu.edu/etd_thesis/2210.

Davis, C.J. 1998. Western pond turtle (Clemmys marmorata). Master's Thesis. Paper 1694. http://scholarworks.sjsu.edu/etd_thesis/1694.

Germano, D.J. 2010. Ecology of western pond turtles (*Actinemys marmorata*) at sewage-treatment facilities in the San Joaquin Valley, California. The Southwestern Naturalist, 55(1): 89-97.

Germano, D.J. and Rathbun, G.B. 2008. Growth, population structure, and reproduction of western pond turtles (*Actinemys marmorata*) on the central coast of California. Chelonian Conservation and Biology 7(2): 188-194.

Lovich, J. and Meyer, K. 2002. The western pond turtle (*Clemmys marmorata*) in the Mojave River, California, USA: highly adapted survivor or tenuous relict? Journal of Zoology London 256: 537-545.

Lubcke, G.M. and Wilson, D.S. 2007. Variation in shell morphology of the western pond turtle (*Actinemys marmorata* Baird and Girard) from three aquatic habitats in northern California. Journal of Herpetology 41(1): 107-114.

Polo-Cavia, N., Engstrom, T., Lopez, P., and Martin, J. 2010. Body condition does not predict immunocompetence of western pond turtles in altered versus natural habitats. Animal Conservation 13: 256-264.

Scott, N.J., Rathbun, G.B., Murphy, T.G., and Harker, M.B. 2008. Reproduction of pacific pond turtles (*Actinemys marmorata*) in coastal streams of central California. Herpetological Conservation and Biology 3(2): 143-148.

Spinks, P.Q. and Shaffer, H.B. 2005. Range-wide molecular analysis of the western pond turtle (*Emys marmorata*): cryptic variation, isolation by distance, and their conservation implications. Molecular Ecology 14: 2047-2064.

Vander Haegen, W.M., Clark, S.L., Perillo, K.M., Anderson, D.P., and Allen, H.L. 2009. Survival and causes of mortality of head-started western pond turtles on Pierce National Wildlife Refuge, Washington. The Journal of Wildlife Management 73(8): 1402-1406.

Foothill Yellow-legged Frog

Bondi, C. A., S.M. Yarnell, and A. J. Lind. 2013. Transferability of habitat suitability criteria for a stream breeding frog (Rana boylii) in the Sierra Nevada, California. Herpetoloty Conservation and Biology 8(1): 88-103.

Catenazzi, A. and S. J. Kupferberg. 2013. The importance of thermal conditions to recruitment success in stream-breeding frog populations distributed across a productivity gradient. Biological Conservation 168: 40-48.

Dever, J.A. 2007. Fine-scale genetic structure in the threatened foothill yellow-legged frog (Rana boylii). Journal of Herpetology 41(1): 168-173.

Fellers, G.M. 2005. Rana boylii Amphibiaweb account. In: Amphibiaweb <www.amphibiaweb.org>. Downloaded on 26 June 2012.

Hayes, Marc P.; Wheeler, Clara A.; Lind, Amy J.; Green, Gregory A.; Macfarlane, Diane C., tech. coords. 2016. Foothill yellow-legged frog conservation assessment in California. Gen. Tech. Rep. PSW-GTR-248. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 193 p.

Jennings, M. R. and Hayes, M. P. 1994. Foothill yellow-legged frog Rana boylii Baird 1854. In: Amphibian and Reptile Species of Special Concern in California. California Department of Fish and Game, Sacramento, California.

Kupferberg, S.J. 1996. Hydrologic and geomorphic factors affecting conservation of a river-breeding frog (Rana boylii). Ecological Applications. 6: 1332–1344.

Kupferberg, S.; Lind, A.; Yarnell, S.; Mount, J. 2008. Pulsed flow effects on the foothill yellow-legged frog (Rana boylii): integration of empirical, experimental and hydrodynamic modeling approaches. Final report. Sacramento, CA: California Energy Commission, Public Interest Energy Research Program. 194 p.

Kupferberg, S.J.; Catenazzi, A.; Power, M.E. 2011. The importance of water temperature and algal assemblage for frog conservation in northern california rivers with hydroelectric projects. CEC-500-2014-033. Sacramento, CA: California Energy Commission, Public Interest Energy Research Program. 96 p.

Leidy, R.A. Gonsolin, E. and Leidy, G.A. 2009. Late-summer aggregation of the foothill yellow-legged frog (Rana boylii) in central California. The Southwestern Naturalist 54(3): 367-368.

Lind, A.J., Spinks, P.Q., Fellers, G.M. and Shaffer, H.B. 2011. Rangewide phylogeography and landscape genetics of the Western U.S. endemic frog Rana boylii (Ranidae): implications for the conservation of frogs and rivers. Conservation Genetics 12: 269-284.

Lind, A.J., and B.H. Sheffer. 2005. Reintroduction of a declining amphibian: Determining an ecologically feasible approach for the foothill yellow-legged frog (Rana boylii) through analysis of decline factors, genetic structure, and habitat associations. Dissertation Abstracts International. Section B: Physical Sciences and Engineering: 66(4) 1853.

Lind, A.J., Welsh Jr., H.H. and Wilson, R.A. 1996. The effects of a dam on breeding habitat and egg survival of the foothill yellow-legged frog (Rana boylii) in northwestern California. Herpetological Review 27(2): 62-67.

Loomis,R.B. 1965. The yellow-legged frog, Rana boylii, from the Sierra San Pedro Martir, Baja California Norte, Mexico. Herpetologica 21(1): 78-80.

Kupferberg, S.J. 1996. Hydrologic and geomorphic factors affecting conservation of a river-breeding frog (Rana boylii). Ecological Applications 6(4): 1332-1344.

Kupferberg, S.J., Catenazzi, A., Lunde, K., Lind, A.J., and Palen, W.J. 2009. Parasitic copepod (Lernaea cyprinacea) outbreaks in foothill yellow-legged frogs (Rana boylii) linked to unusually warm summers and amphibian malformations in northern California. Copeia 2009(3): 529-537.

Kupferberg, S.J., Lind, A.J., Thill, V. and Yarnell, S.M. 2011. Water velocity tolerance in tadpoles of the foothill yellow-legged frog (Rana boylii): swimming performance, growth, and survival. Copeia 2011(1): 141-152.

Morey, S. and Papenfuss, T. 1990. Foothill yellow-legged frog Rana boylii. In: California's Wildlife. Vol. I. Amphibians and Reptiles. D. C. Zeiner, W. F. Laudenslayer Jr., K. E. Mayer, and M. White. California Department of Fish and Game, Sacramento, California.

Paoletti, D.J., Olson, D.H. and Blaustein, A.R. 2011. Responses of foothill yellow-legged frog (Rana boylii) larvae to an introduced predator. Copeia 2011(1): 161-168.

Rombough, C.J. and Hayes, M.P. 2005. Novel aspects of oviposition site preparation by foothill yellow-legged frogs (Rana boylii). Northwestern Naturalist 86: 157-160.

Santos-Barrera, G., Hammerson, G. and Fellers, G. 2004. Rana boylii. In: IUCN 2012. IUCN Red List of the Threatened Species. Version 2012.1. <www.iucnredlist.org>. Downloaded on 26 June 2012.

Sparling,D.W. and Fellers, G. 2007. Comparative toxicity of chlorpyrifos, diazinon, malathion and their oxon derivatives to larval Rana boylii. Environmental Pollution 147: 535-539.

Wheeler, C.A. and Welsh Jr., H.H. 2007. Mating strategy and breeding patterns of the foothill yellow-legged frog (Rana boylii). Herpetological Conservation and Biology 3(2): 128-142.

Wheeler, C.A.; Bettaso, J.B.; Ashton, D.T.; Welsh, H.H., Jr. 2013. Effects of water temperature on breeding phenology, growth and timing of metamorphosis of foothill yellow-legged frogs (Rana boylii) on the mainstem and selected tributaries of California's Trinity River—2004–2009. Report to the Trinity River Restoration Program. Arcata, CA: U.S. Department Agriculture, Pacific Southwest Research Station and U.S. Fish and Wildlife Service. 37 p.

Sierra Nevada (Mountain) Yellow-legged Frog

Bahls, P. 1992. The status of fish populations and management of high mountain lakes in the western United States. Northwest Science 66(3):183-193.

Berger, L., R. Speare, P. Daszak, D. E. Green, A. A. Cunningham, C. L. Goggin, R. Slocombe, M. A. Ragan, A. D. Hyatt, K. R. McDonald, H. B. Hines, K. R. Lips, G. Marantelli, and H. Parkes. 1998. Chytridiomycosis causes amphibian mortality associated with population declines in the rain forests of Australia and Central America. Proceedings of the National Academy of Sciences USA 95:9031-9036.

Berger, L., G. Marantelli, L. F. Skerratt, and R. Speare. 2005. Virulence of the amphibian chytrid fungus *Batrachochytrium dendrobatidis* varies with the strain. Diseases of Aquatic Organisms 68:47-50.

Bosch, J., I. Martínez-Solano, and M. García-París. 2001. Evidence of a chytrid fungus infection involved in the decline of the common midwife toad (Alytes obstetricans) in protected areas of central Spain. Biological Conservation 97:331-337.

Bradford, D. F. 1983. Winterkill, oxygen relations, and energy metabolism of a submerged dormant amphibian, *Rana muscosa*. Ecology 64(5):1171-1183.

Bradford, D. F. 1984. Temperature modulation in a high-elevation amphibian, *Rana muscosa*. Copeia 1984(4):966-976.

Bradford, D. F. 1989. Allotropic distribution of native frogs and introduced fishes in high Sierra Nevada lakes of California: implication of the negative effect of fish introductions. Copeia 1989(3):775-778.

Bradford, D. F. 1991. Mass mortality and extinction in a high elevation population of *Rana muscosa.* Journal of Herpetology 25(2):174-177.

Bradford, D.F., Graber, D.M., and Tabatabai, F. 1994. Population declines of the native frog, Rana muscosa, in Sequoia and Kings Canyon National Parks, California. The Southwestern Naturalist 39(4): 323-327.

Bradford, D. F., F. Tabatabai, and D. M. Graber. 1993. Isolation of remaining populations of the native frog, *Rana muscosa*, by introduced fishes in Sequoia and Kings Canyon National Parks, California. Conservation Biology 7: 882-888.

Bradford, D.F., R. A. Knapp, D.W. Sparling, M.S. Nash, K.A. Stanley, , N.G. Tallent-Halsell, L.L. McConnell, and S.M. Simonich. 2011. Pesticide distributions and population declines of California, USA, alpine frogs, Rana muscosa and Rana sierrae. Environmental Toxicology and Chemistry 30(3): 682-691

Bradley, G. A., P. C. Rosen, M. J. Sredl, T. R. Jones, and J. E. Longcore. 2002. Chytridiomycosis in native Arizona frogs. Journal of Wildlife Diseases 38(1):206-212.

Briggs, C. J., V. T. Vredenburg, R. A. Knapp, and L. J. Rachowicz. 2005. Investigating the population-level effects of chytridiomycosis: an emerging infectious disease of amphibians. Ecology 86 (12):3149-3159.

Carey, C., N. Cohen, and L. Rollins-Smith. 1999. Amphibian declines: an immunological perspective. Developmental and Comparative Immunology 23:459-472.

Davidson, C. 2004. Declining downwind: amphibian population declines in California and historical pesticide use. Ecological Applications 14(6):1892-1902.

Davidson, C., and R. A. Knapp. 2007. Multiple stressors and amphibian declines: dual impacts of pesticides and fish on yellow-legged frogs. Ecological Applications 17(2):587-597.

Drost, C. A., and G. M. Fellers. 1996. Collapse of a regional frog fauna in the Yosemite area of the California Sierra Nevada, USA. Conservation Biology 10(2):414-425.

Fellers, G. M., D. E. Green, and J. E. Longcore. 2001. Oral chytridiomycosis in the mountain yellow-legged frog (*Rana muscosa*). Copeia 2001(4):945-953.

Fellers, G.M., L.L. McConnell, D. Pratt, and, S. Datta. 2004. Pesticides in mountain yellow-legged frogs (*Rana muscosa*) from the Sierra Nevada mountains of California, USA. Environmental Toxicology and Chemistry 23(9): 2170-2177.

Fellers, G. M., D. F. Bradford, D. Pratt, and L. Long Wood. 2007. Demise of repatriated populations of mountain yellow-legged frogs (*Rana muscosa*) in the Sierra Nevada of California. Herpetological Conservation and Biology 2(1):5-21.

Finlay, J. C., and V. T. Vredenburg. 2007. Introduced trout sever trophic connections in watersheds: consequences for a declining amphibian. Ecology 88(9):2187-2198.

Hayes, M. P., and M. R. Jennings. 1986. Decline of ranid frog species in western North America: are bullfrogs (*Rana catesbeiana*) responsible? Journal of Herpetology 20(4):490-509.

Jennings, M. 1993. Current Status of 25 species of California Herps. Presented at the 1993 Wildlife Society Western Section Meeting.

Jennings, M. R. and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. California Department of Fish and Game, Inland Fisheries Division, Rancho Cordova, California.

Knapp, R. A. 1993. Non-native trout in natural lakes of the Sierra Nevada: An analysis of their distribution and impacts on native aquatic biota. Sierra Nevada Ecosystem Project: Final Report to Congress, Volume III, Chapter 8, pp 363-407. Davis: University of California, Centers for Water and Wildland Resources.

Knapp, R. 1994. Wilderness Record. Volume 19, Number 2. Davis, California.

Knapp, R. A., and K. R. Matthews. 2000. Non-native fish introductions and the decline of the mountain yellow-legged frog from within protected areas. Conservation Biology 14(2):428-438.

Knapp, R. A., K. R. Matthews, H. K. Preisler, and R. Jellison. 2003. Developing probabilistic models to predict amphibian site occupancy in a patchy landscape. Ecological Applications 13(4):1069-1082.

Knapp, R. A., D. M. Boiano, and V. T. Vredenburg. 2007. Removal of nonnative fish results in population expansion of a declining amphibian (mountain yellow-legged frog, Rana muscosa). Biological Conservation 135:11-20.

Knapp, R.A., C.J. Briggs, T.C. Smith, and J.R. Maurer. 2011. Nowhere to hide: impact of a temperature-sensitive amphibian pathogen along an elevation gradient in the temperate zone. Ecosphere 2(8): 1-25

Lacan, I., K. Matthews, and K. Feldman. 2008. Interaction of an introduced predator with future effects of climate change in the recruitment dynamics of the imperiled Sierra Nevada yellow-legged frog (Rana sierrae). Herpetological Conservation and Biology 3(2): 211-223.

Lam, B.A., J.B. Walke, V.T. Vredenburg and R.N. Harris. 2010. Proportion of individuals with anti-Batrachochytrium dendrobatidis skin bacteria is associated with population persistence in the frog Rana muscosa. Biological Conservation 143: 529-531.

Longcore, J. E., A. P. Pessier, and D. K. Nichols. 1999. *Batrachochytrium dendrobatidis* gen. et sp. nov., a chytrid pathogenic to amphibians. Mycologia 91:219-227.

Macey, J. R., J. L. Strasburg, J. A. Brisson, V. T. Vredenburg, M. Jennings, and A. Larson. 2001. Molecular phylogenetics of western North American frogs of the *Rana boylii* species group. Molecular Phylogenetics and Evolution 19(1):131-143.

Martin, D. L. 1992. Sierra Nevada Anuran Guide. Canorus, LTD. Ecological Research Team. San Jose, CA.

Martin, D. L. 1994. Personal communication regarding mountain yellow-legged frog habitat needs.

Matthews, K. R. 2003. Response of mountain yellow-legged frogs, *Rana muscosa*, to short distance translocation. Journal of Herpetology 37(3):621-626.

Matthews, K. R. and C. Miaud. 2007. A skeletochronological study of the age structure, growth, and longevity of the mountain yellow-legged frog, *Rana muscosa*, in the Sierra Nevada, California. Copeia 2007(4): 986-993.

Matthews, K. R., and K. L. Pope. 1999. A telemetric study of the movement patterns and habitat use of *Rana muscosa*, the mountain yellow-legged frog, in a high-elevation basin in Kings Canyon National Park, California. Journal of Herpetology 33(4):615-624.

Matthews, K. R. and Preisler, H. K. 2010. Site fidelity of the declining amphibian Rana sierrae (Sierra Nevada yellow-legged frog). Canadian Journal of Fisheries and Aquatic Science 67: 243-255.

Mullally, D. P. and J. D. Cunningham. 1956. Ecological relations of Rana muscocsa at high elevations in the Sierra Nevada. Herpetologica 12(3): 189-198

Muths, E., P. S. Corn, A. P. Pessier, and D. E. Green. 2003. Evidence for disease-related amphibian decline in Colorado. Biological Conservation 110:357-365.

Oullet, M., I. Mikaelian, B. D. Pauli, J. Rodrigue, and D. M. Green. 2005. Historical evidence of widespread chytrid infection in North American amphibian populations. Conservation Biology 19(5):1431-1440.

Pope, K. L., and K. R. Matthews. 2001. Movement ecology and seasonal distribution of mountain yellow-legged frogs, *Rana muscosa*, in a high-elevation Sierra Nevada basin. Copeia 2001(3):787-793.

Pope, K. L., and K. R. Matthews. 2002. Influence of anuran prey on the condition and distribution of Rana muscosa in the Sierra Nevada. Herpetologica 58(3):354-363.

Rachowicz, L. J., and V. T. Vredenburg. 2004. Transmission of *Batrachochytrium dendrobatidis* within and between amphibian life stages. Diseases of Aquatic Organisms 61:75-83.

Rachowicz, L. J., R. A. Knapp, J. A. T. Morgan, M. J. Stice, V. T. Vredenburg, J. M. Parker, and C. J. Briggs. 2006. Emerging infectious disease as a proximate cause of amphibian mass mortality. Ecology 87(7):1671-1683.

Rachowicz, L. J., and C. J. Briggs. 2007. Quantifying the disease transmission function: effects of density on *Batrachochytrium dendrobatidis* transmission in the mountain yellow-legged frog *Rana muscosa*. Journal of Animal Ecology 76:711-721.

Schoville, S. D., T. S. Tustall, V. T. Vredenburg, A. R. Backlin, E. Gallegos, D. A. Wood, and R. N. Fisher. 2011. Conservation genetics of evolutionary lineages of the endangered mountain yellow-legged frog, Rana muscosa (Amphibia: Ranidae), in southern California. Biological Conservation 144: 2031-2040.

Woodhams, D.C., V. T. Vredenburg, M. A. Simon, D. Billheimer, B. Shakhtour, Y. Shyr, C. J. Briggs, L. A. Rollins-Smith, and R. N. Harris. 2007. Symbiotic bacteria contribute to innate immune defenses of the threatened mountain yellow-legged frog, Rana muscosa. Biological Conservation 138: 390-398

U. S. Fish and Wildlife Service (USFWS) 2000. Endangered and Threatened Wildlife and Plants: 90-day Finding on a Petition to List Mountain Yellow-legged Frog as Endangered. Federal Register 65(198):60603-60605.

U. S. Fish and Wildlife Service (USFWS). 2002. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for the Southern California Distinct Vertebrate Population Segment of the Mountain Yellow-Legged Frog (*Rana muscosa*). Federal Register 67(127):44382-44392.

U. S. Fish and Wildlife Service (USFWS). 2003. Endangered and Threatened Wildlife and Plants; 12-Month Finding for a Petition To List the Sierra Nevada Distinct Population Segment of the Mountain Yellow-legged Frog (*Rana muscosa*). Federal Register 68(11):2283-2303.

U. S. Fish and Wildlife Service (USFWS). 2006. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for the Southern California Distinct Population Segment of the Mountain Yellow-Legged Frog (*Rana muscosa*). Federal Register 71(127):54344-54386.

U. S. Fish and Wildlife Service (USFWS). 2007. Endangered and Threatened Wildlife and Plants; 12-Month Finding on a Petition To List the Sierra Nevada Distinct Population Segment of the Mountain Yellow-Legged Frog (*Rana muscosa*). Federal Register 72(121):34657-34661.

U. S. Fish and Wildlife Service (USFWS). 2014. Endangered and Threatened Wildlife and Plants; Endangered Species Status for Sierra Nevada Yellow-Legged Frog and Northern Distinct Population Segment of the Mountain Yellow-Legged Frog, and Threatened Species Status for Yosemite Toad. Federal Register (Final Rule) 79:24255-24310.

U.S. Fish and Wildlife Service (USFWS). 2016. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for the Sierra Nevada Yellow-Legged Frog, the Northern DPS of the Mountain Yellow-Legged Frog, and the Yosemite Toad. Federal Register (Final Rule) 81(166): 59046-59119.

Vindum, J. V., M. M. Tierney, and M. S. Koo. 1997. 1997 Amphibian and reptile surveys in the Tahoe National Forest: the result of CCSA-05-97-17-024. Department of Herpetology, California Academy of Sciences, San Francisco, California.

Vindum, J. V., and M. S. Koo. 1999a. 1998 Amphibian and reptile surveys in the Plumas and Tahoe National Forests: the result of CCSA-05-98-17-123. Department of Herpetology, California Academy of Sciences, San Francisco, California.

Vindum, J. V., and M. S. Koo. 1999b. Amphibians and reptiles of the Tahoe National Forest: historical collections and results of 1997-1999 California Academy of Sciences surveys. Department of Herpetology, California Academy of Sciences, San Francisco, California.

Voyles, J., S. Young, L. Berger, C. Campbell, W. F. Voyles, A. Dinudom, D. Cook, R. Webb, R. A. Alford, L. F. Skerratt, and R. Speare. 2009. Pathogenesis of chytridiomycosis, a cause of catastrophic amphibian declines. Science 326:582-585.

Vredenburg, V.T. 2004. Reversing introduced species effects: experimental removal of introduced fish leads to rapid recovery of a declining frog. Proceedings of the Natural Academy of Sciences 101(20):7646-7650.

Vredenburg, V., G. M. Fellers, and C. Davidson. 2005. The mountain yellow-legged frog Rana muscosa (Camp 1917). Pp. 563-566 *In* M. Lannoo (Editor); Amphibian Declines: The Conservation Status of United States Species. University of California Press, Berkeley, California.

Vredenburg, V. T., R. Bingham, R. Knapp, J. A. T. Morgan, C. Moritz, D. Wake. 2007. Concordant molecular and phenotypic data delineate new taxonomy and conservation priorities for the endangered mountain yellow-legged frog. Journal of Zoology 271: 361-374.

Wake, D. B., and V. T. Vredenburg. 2008. Are we in the midst of the sixth mass extinction? A view from the world of amphibians. Proceedings of the National Academy of Sciences 105(Suppl. 1):11466-11473.

Woodhams, D. C., R. A. Alford, and G. Marantelli. 2003. Emerging disease of amphibians cured by elevated body temperature. Diseases of Aquatic Organisms 55:65-67.

Woodhams, D. C., K. Ardipradja, R. A. Alford, G. Marantelli, L. K. Reinert, and L. A. Rollins-Smith. 2007. Resistance to chytridiomycosis varies among amphibian species and is correlated with skin peptide defenses. Animal Conservation 10:409-417.

Zeiner, D.C., W.F. Laudenslayer, Jr., K. E. Mayer, and M. White. 1988. California's Wildlife. Volume I. Amphibians and Reptiles. California Statewide Wildlife Habitat Relationships System. Department of Fish and Game, The Resources Agency, Sacramento, CA. 272 pages.

Great Basin Rams-horn Snail

Frest, T. J. and E. J. Johannes. 1993. Mollusk Species of Special Concern Within the Range of the Northern Spotted Owl. Final Report to the Forest Ecosystem Management Working Group, USDA, Forest Service. Deixis Consultants, Seattle WA. 98 pages.

Taylor, D. W. 1981. Freshwater Mollusks of California: A Distributional Checklist. California Department of Fish and Game 67(3):140-163.

Lahontan Lake Tui Chub

Marrin, D. L. and D. C. Erman. 1982. Evidence against competition between trout and nongame fish in Stampede Reservoir, California. No. Amer. J. Fish. Manage. 2:262-269.

Miller, R. G. 1951. The natural history of Lake Tahoe fishes. Unpubl. Dissertation. Stanford University. 160 pp.

Moyle, P. B. 1976. Inland Fishes of California. University of California Press, Berkeley. 405pp.

Moyle, P. B., R. M. Yoshiyama, J. E. Williams, and E. D. Wikramanyake. 1995. Fish species of special concern in California. Dept. Fish and Game, Inland Fisheries Division. Rancho Cordova, CA Final Report for contract No. 2128IF. 272 pp.

Richards, R. C., C. R. Goldman, T. C. Frantz, and R. Wickwire. 1975. Where have all the *Daphnia* gone? The decline of a major cladoceran in Lake Tahoe, California-Nevada. Verh. Internat. Verein. Limnol. 19:835-842.

Snyder, J. O. 1917. Fishes of the Lahontan system of Nevada and northeastern California. Bull. U.S. Bur. Fish. 35:31-86.

Hardhead

Avise, J. C., and F. J. Ayala. 1976. Genetic differentiation in species versus depauperate phylads: Evidence from the California minnows. Evolution 30: 46-58.

Ayres, W. O. 1854. Remarks on some fish of the cyprinoid family. Daily Placer Times and Transcripts, 30 May 1854:1.

Baltz, D. M. and P. B. Moyle. 1993. Invasion resistance by an assemblage of native California stream fishes. Ecol. Applic. 3:246-255.

Bell, R. R. and J. B. Kimsey. 1955. Some considerations of the chemical treatment of the Monticello Reservoir basin, Napa County. Calif. Dept. of Fish and Game, Inland Fish. Admin. Rep. No. 55. 7pp.

Brown, L. R. and P. B. Moyle. 1987. Survey of fishes of mid-elevation streams of the San Joaquin Valley. Unpubl. Rep. Calif. Dept. of Fish and Game. 220pp.

Cech, J. J., S. J. Mitchell, D. T. Castleberry, and M. McEnroe. 1990. Distribution of California stream fishes: influence of environmental temperature and hypoxia. Env. Biol. Fish. 29:95-105.

Cooper, J. J. 1983. Distributional ecology of native and introduced fishes in the Pit River system, northeastern California, with note on the Modoc sucker. Calif. Fish and Game 69:39-53.

Evermann, B. W. 1905. The golden trout of the southern High Sierras. Bull. U.S. Bur. Fish. 25:1-51.

Girard, C. 1854. Description of *Sebastues ruber, Sebastes ruber* var. *parvu, Sebastes variabilis*, and *Centrarchus maculosus*. The Pacific 3: 165-209.

Jordan, D. S. and B. W. Evermann. 1896. Fishes of North and Middle America. Bull. U.S. Nat. Mus. 47:1-3705.

Knight, N. J. 1985. Microhabitats and temperature requirements of hardhead (*Mylopharadon conocephalus*) and Sacramento squawfish (*Ptychocheilus grandis*), with notes for some other native California stream fishes. Unpubl. Ph.D Dissertation., University of California, Davis. 161pp.

Moyle, P. B. 1976. Inland Fishes of California. University of California Press, Berkeley. 405pp.

Moyle, P. B. and D. M. Baltz. 1985. Microhabitat use by an assemblage of California stream fishes: developing criteria for instream flow recommendations. Trans. Amer. Fish. Soc. 114:695-704.

Moyle, P. B. and R. A. Daniels. 1982. Fishes of the Pit River system, and Surprise Valley region. University of Calif. Publ. Zool. 115:1-82.

Moyle, P. B. and M. Gard. 1993. South Yuba River fisheries survey and habitat assessment. Report to the California State Department of Parks and Recreation. 66pp.

Moyle, P. B. and R. D. Nichols. 1973. Ecology of some native and introduced fishes of the Sierra Nevada foothills, central California. Copeia. 1973: 478-490.

Moyle, P. B., B. Vondracek and G. D. Grossman. 1983 Response of fish populations in the North Fork of the Feather River, California to treatment with fish toxicants. N. Amer. J. Fish. Mgmt. 3:48-60.

Murphy, G. I. 1947. Notes on hardhead control in the East Fork of the Russian River, Mendocino County, California. Calif. Dept. Fish and Game. 34:101-110.

Pacific Gas and Electric (PG&E). 1985. Bald Eagle and Fish Study. Pit 3,4 and 5 Project. Biosystems Analysis, Inc. and University of California, Davis.

Reeves, J. G. 1964. Age and growth of the hardhead minnow *Mylopharodon conocephalus* in the American River basin of California, with notes on its ecology. M.S. Thesis. University of California, Berkeley. 90 pp.

Rowley, W. 1955. 1954 creel census, North Fork, Feather River. Calif. Dept. Fish and Game, Inland Fish. Admin. Rep. 55 17pp.

Rutter, C. 1908. The fishes of the Sacramento-San Joaquin basin, with a study of their distribution and variation. U.S. Bur. Fish. Bull. 27:143-48.

Saiki, M. K. 1984. Environmental conditions and fish faunas in low elevation rivers an the irrigated San Joaquin Valley floor, California. Calif. Fish and Game 70:145-157

Soule, M. 1951. Power development of the Kings River drainage, Fresno County, California. Rep. No 3., Junction development of the Kings River and its Middle and South Forks. Calif. Dept. Fish and Game, Inland Fish. Admin. Rep. 51. 17pp.

Wales, J. H. 1946. The hardhead problem in the Sacramento River above Shasta Lake, California. Calif. Dept. Fish and Game, Inland fish. Admin. Rep. 46. 4pp.

Wang, J. C. S. 1986. Fishes of the Sacramento-San Joaquin estuary and adjacent waters, California: A guide to the early life histories. Interagency Ecological Study Program for the Sacramento- San Joaquin Estuary, Tech. Rep. 9.

California Floater

Brim Box, J., S. Chappell, M. McFarland and J. Furnish. 2005. The aquatic mollusk fauna of the Lassen National Forest in northeastern California. Draft report for USDA/Forest Service, Rocky Mountain Research Station, Logan, Utah.

Chong, J.P., J. Brim Box, J. Howard, D. Wolf, T. Myers, K. Mock. 2008. Three deeply divided lineages of the freshwater mussel genus Anodonta in western North America. Conservation Genetic 9:1303-1309.

David, A.. 2008. Abundance, Diversity and Distribution of Freshwater Mussels (Bivalvia: Unionoidea) in the Klamath and Salmon Rivers of Northern California. Report prepared for the Biology and Environmental Studies Departments, Whitman College, Walla Walla, Washington.

Ellis, M. 2003. Effects of High Test Flows on the Malinda Gulch Mussel Bed in Pacific Gas and Electric Company's Pit 3, 4, and 5 Hydroelectric Project (FERC No. 233). A final report prepared by Spring Rivers Ecological Sciences, Cassel, CA for PGE, San Ramon, CA. 19 pp.

Ellis, M. J. and L. E. Haley. 2005. Reproductive timing of freshwater mussels and the potential impacts of pulsed flows on reproductive success, Spring Rivers Ecological Sciences, Cassel, CA. Paper presented at the Pulsed Flow Conference, July 15, 2005, Davis, CA.

Frest, T. J. and E. J. Johannes. 1995. Freshwater mollusks of the upper Sacramento system, California, with particular reference to the Cantara Spill. 1994 yearly report by Deixis Consultants (Seattle, WA) to California Department of Fish and Game, iii+88 pages and appendices, March 13.

Hovingh, P. 2004. Intermountain freshwater mollusks, USA (Margaritifera, Anodonta, Gonidea, Valvata, Ferrissia): geography, conservation, and fish management implications. Monographs of the Western North American Naturalist 2:109-135.

Howard, J. 2008. Strategic inventory of freshwater mussels in the northern Sierra Nevada Province. Draft Progress Report. Western Mollusk Sciences, San Francisco, CA.

Howard, J. and K. Cuffey, 2003. Freshwater mussels in a California North Coast Range river: occurrence, distribution, and controls. Journal of the North American Benthological Society. 22: 63-77.

Mock, K. E., J. C. Brim-Box, M. P. Miller, M. E. Downing and W. R. Hoeh. 2004. Genetic diversity and divergence among freshwater mussel (Anodonta) populations in the Bonneville Basin of Utah. Molecular Ecology 13:1085-1098.

Taylor, D. W. 1981. Freshwater mollusks of California: A distributional checklist. California Fish and Game 67: 140-163.

Black Juga

Barquín, J. and M. Scarsbrook. 2008. Management and conservation strategies for coldwater springs. Aquatic Conservation: Marine and Freshwater Ecosystems 18: 580–591. Available at http://onlinelibrary.wiley.com/doi/10.1002/aqc.884/pdf.

Burch, J.B. 1989. North American Freshwater Snails. Malacological Publications (J. B. Burch, Hamburg, Michigan), viii+365 pages.

Campbell, D., T. Frest, S.A. Clark, E. Johannes and C. Lydeard. 2010 Molecular evidence on the phylogeny of the freshwater gastropod Juga (Gastropoda: Cerithiodea:Pleuroceridae) for the northwestern United States. Manuscript submitted for publication to The Nautilus.

Frest, T. J. and E. J. Johannes. 1995. Freshwater Mollusks of the Upper Sacramento System, California, with Particular Reference to the Cantara Spill. 1994 Yearly report to California Department of Fish & Game. Deixis Consultants, Seattle, Washington. Contract #FG2106R1.

Furnish, J.L.. 1990. Factors affecting the growth, production and distribution of the stream snail Juga silicula (Gould). Oregon State University (Corvallis), unpublished doctoral dissertation. 173 pp.

Furnish, J.L. and R. Monthey. 1999. Management recommendations for Survey and Manage aquatic mollusks, version 2.0, December 1998. USDA Forest Service (Regions 5 and 6) and USDI Bureau of Land Management (Oregon, Washington, California), [124 pages], distributed 3 March 1999. Available at http://www.blm.gov/or/plans/surveyandmanage/MR/AQMollusks/section4.htm

Henderson, J. 1929. The non-marine Mollusca of Oregon and Washington. University of Colorado (Boulder) Studies 17(2): 47-190.

Holznagel, E.E. and C. Lydeard. 2000. A molecular phylogeny of North American Pleuroceridae (Gastropoda:Cerithioidea) based on mitochondrial 16S rDNA sequences. Journal of Molluscan Studies 66:233-257.

Johannes, E. 2010. Personal communication, Deixis Consultants, December, 2010.
NatureServe Explorer online: www.natureserve.org/explorer

Sada, D.W.; Williams, J.E.; Silvey, J.C.; Halford, A.; Ramakka, J.; Summers, P.; Lewis, L. 2001. Riparian area management. A guide to managing, restoring, and conserving springs in the western United States. Technical Reference 1737-17. Denver, CO. Available at http://digitalcommons.usu.edu/govdocs/252/ .

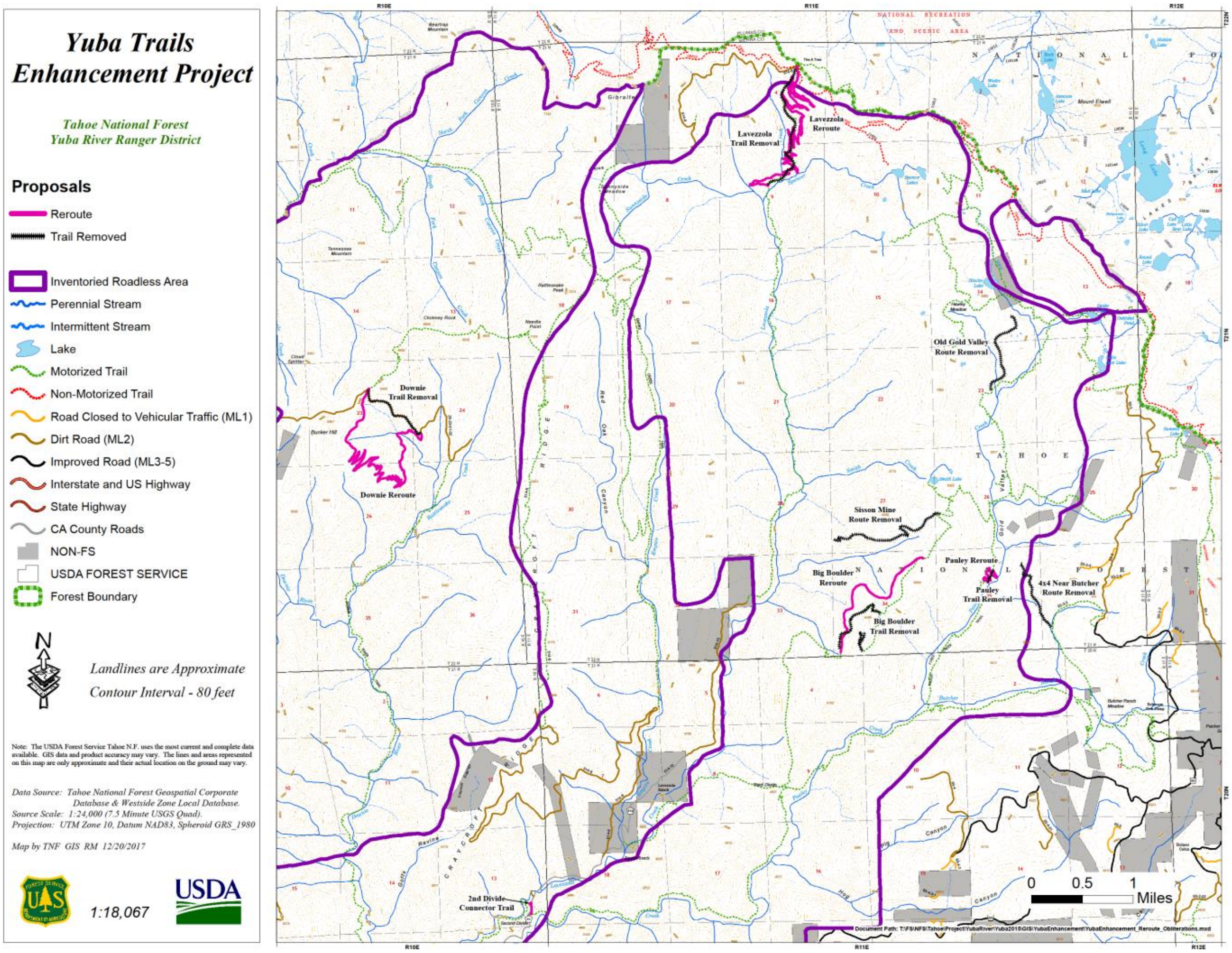
Sada, D. W. and K.F. Pohlmann. 2002. Spring Inventory and Monitoring ProtocolsConference Proceedings. Spring-fed Wetlands: Important Scientific and Cultural Resources of the Intermountain Region, 2002. Available at http://www.wetlands.dri.edu/2002/Sada_Pohlmann_Protocol.pdf

Schmitz, D., B. McGlynn and D. Patten. 2007. Development of a spring ecosystem monitoring protocol in Bighorn Canyon National Recreation Area. Final Report Submitted to the Greater Yellowstone Network Inventory and Monitoring Program National Park Service from the Land Resources and Environmental Sciences Department, Montana State University, Bozeman, MT. Available at http://www.cfc.umt.edu/cesu/NEWCESU/Assets/Individual%20Project%20Reports/NPS%20Projects/MSU/2004/McGlynn_seeps_rpt.pdf.

Strong, E.E. O. Gargominy, W.F. Ponder and P. Bouchet. 2008. Global diversity of gastropods (Gastropoda: Mollusca) in freshwater. Hydrobiologia 595:149-166. Avaialbe at http://inpn.mnhn.fr/gargo/strong%20et%20al_2008.pdf .

Taylor, D. W. 1981. Freshwater mollusks of California: A distributional checklist. California Fish and Game 67: 140-163.

Figure 1.


Yuba Trails Enhancement Project
Tahoe National Forest
Yuba River Ranger District
Proposals
Reroute
Trail Removed
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Perennial Stream
Intermittent Stream
Lake
Motorized Trail
Non-Motorized Trail
Road Closed to Vehicular Traffic (ML1)
Dirt Road (ML2)
Improved Road (ML3-5)
Interstate and US Highway
State Highway
CA County Roads
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USDA FOREST SERVICE
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N
Landlines are Approximate
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Note: The USDA Forest Service Tahoe N.F. uses the most current and complete data available. GIS data and product accuracy may vary. The lines and areas represented on this map are only approximate and their actual location on the ground may vary.
Data Source: Tahoe National Forest Geospatial Corporate Database & Westside Zone Local Database.
Source Scale: 1:24,000 (7.5 Minute USGS Quad).
Projection: UTM Zone 10, Datum NAD83, Spheroid GRS_1980
Map by TNF GIS RM 12/20/2017
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USDA
Lavezzola Trail Removal
Lavezzola Reroute
Old Gold Valley Route Removal
Downie Trail Removal
Downie Reroute
Sisson Mine Route Removal
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Appendix A.

United States Department of the Interior

FISH AND WILDLIFE SERVICE
Sacramento Fish And Wildlife Office
Federal Building
2800 Cottage Way, Room W-2605
Sacramento, CA 95825-1846
Phone: (916) 414-6600 Fax: (916) 414-6713

In Reply Refer To: August 01, 2018
Consultation Code: 08ESMF00-2015-SLI-0871
Event Code: 08ESMF00-2018-E-08496
Project Name: Tahoe NF TE species list

Subject: Updated list of threatened and endangered species that may occur in your proposed project location, and/or may be affected by your proposed project

To Whom It May Concern:
The enclosed species list identifies threatened, endangered, proposed and candidate species, as well as proposed and final designated critical habitat, under the jurisdiction of the U.S. Fish and Wildlife Service (Service) that may occur within the boundary of your proposed project and/or may be affected by your proposed project. The species list fulfills the requirements of the Service under section 7(c) of the Endangered Species Act (Act) of 1973, as amended (16 U.S.C. 1531 et seq.).
Please follow the link below to see if your proposed project has the potential to affect other species or their habitats under the jurisdiction of the National Marine Fisheries Service:

http://www.nwr.noaa.gov/protected_species/species_list/species_lists.html

New information based on updated surveys, changes in the abundance and distribution of species, changed habitat conditions, or other factors could change this list. Please feel free to contact us if you need more current information or assistance regarding the potential impacts to federally proposed, listed, and candidate species and federally designated and proposed critical habitat. Please note that under 50 CFR 402.12(e) of the regulations implementing section 7 of the Act, the accuracy of this species list should be verified after 90 days. This verification can be completed formally or informally as desired. The Service recommends that verification be completed by visiting the ECOS-IPaC website at regular intervals during project planning and implementation for updates to species lists and information. An updated list may be requested through the ECOS-IPaC system by completing the same process used to receive the enclosed list.

The purpose of the Act is to provide a means whereby threatened and endangered species and the ecosystems upon which they depend may be conserved. Under sections 7(a)(1) and 7(a)(2) of the Act and its implementing regulations (50 CFR 402 et seq.), Federal agencies are required to utilize their authorities to carry out programs for the conservation of threatened and endangered species and to determine whether projects may affect threatened and endangered species and/or designated critical habitat.

A Biological Assessment is required for construction projects (or other undertakings having similar physical impacts) that are major Federal actions significantly affecting the quality of the human environment as defined in the National Environmental Policy Act (42 U.S.C. 4332(2) (c)). For projects other than major construction activities, the Service suggests that a biological evaluation similar to a Biological Assessment be prepared to determine whether the project may affect listed or proposed species and/or designated or proposed critical habitat. Recommended contents of a Biological Assessment are described at 50 CFR 402.12.

If a Federal agency determines, based on the Biological Assessment or biological evaluation, that listed species and/or designated critical habitat may be affected by the proposed project, the agency is required to consult with the Service pursuant to 50 CFR 402. In addition, the Service recommends that candidate species, proposed species and proposed critical habitat be addressed within the consultation. More information on the regulations and procedures for section 7 consultation, including the role of permit or license applicants, can be found in the "Endangered Species Consultation Handbook" at: http://www.fws.gov/endangered/esa-library/pdf/TOC-GLOS.PDF

Please be aware that bald and golden eagles are protected under the Bald and Golden Eagle Protection Act (16 U.S.C. 668 et seq.), and projects affecting these species may require development of an eagle conservation plan (http://www.fws.gov/windenergy/ eagle_guidance.html). Additionally, wind energy projects should follow the wind energy guidelines (http://www.fws.gov/windenergy/) for minimizing impacts to migratory birds and bats.

Guidance for minimizing impacts to migratory birds for projects including communications towers (e.g., cellular, digital television, radio, and emergency broadcast) can be found at: http:// www.fws.gov/migratorybirds/CurrentBirdIssues/Hazards/towers/towers.htm; http:// www.towerkill.com; and http://www.fws.gov/migratorybirds/CurrentBirdIssues/Hazards/towers/ comtow.html.

We appreciate your concern for threatened and endangered species. The Service encourages Federal agencies to include conservation of threatened and endangered species into their project planning to further the purposes of the Act. Please include the Consultation Tracking Number in the header of this letter with any request for consultation or correspondence about your project that you submit to our office.

Attachment(s):

- Official Species List

Official Species List

This list is provided pursuant to Section 7 of the Endangered Species Act, and fulfills the requirement for Federal agencies to "request of the Secretary of the Interior information whether any species which is listed or proposed to be listed may be present in the area of a proposed action".
This species list is provided by:
Sacramento Fish And Wildlife Office
Federal Building
2800 Cottage Way, Room W-2605
Sacramento, CA 95825-1846
(916) 414-6600
This project's location is within the jurisdiction of multiple offices. Expect additional species list documents from the following office, and expect that the species and critical habitats in each document reflect only those that fall in the office's jurisdiction:
Reno Fish And Wildlife Office
1340 Financial Boulevard, Suite 234
Reno, NV 89502-7147
(775) 861-6300

Project Summary

Consultation Code: 08ESMF00-2015-SLI-0871
Event Code: 08ESMF00-2018-E-08496
Project Name: Tahoe NF TE species list
Project Type: VEGETATION MANAGEMENT
Project Description: Projects within Tahoe NF boundary
Project Location:

Approximate location of the project can be viewed in Google Maps: https://www.google.com/maps/place/39.39591036852921N120.57539032363536W

Counties: El Dorado, CA | Lassen, CA | Nevada, CA | Placer, CA | Plumas, CA | Sierra, CA | Yuba, CA

Endangered Species Act Species

There is a total of 11 threatened, endangered, or candidate species on this species list.

Species on this list should be considered in an effects analysis for your project and could include species that exist in another geographic area. For example, certain fish may appear on the species list because a project could affect downstream species.
IPaC does not display listed species or critical habitats under the sole jurisdiction of NOAA Fisheries[1], as USFWS does not have the authority to speak on behalf of NOAA and the Department of Commerce.
See the "Critical habitats" section below for those critical habitats that lie wholly or partially within your project area under this office's jurisdiction. Please contact the designated FWS office if you have questions.

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1. NOAA Fisheries, also known as the National Marine Fisheries Service (NMFS), is an office of the National Oceanic and Atmospheric Administration within the Department of Commerce.

Birds

NAME STATUS

Yellow-billed Cuckoo Coccyzus americanus Threatened
Population: Western U.S. DPS
There is proposed critical habitat for this species. Your location is outside the critical habitat.
Species profile: https://ecos.fws.gov/ecp/species/3911

Amphibians

NAME STATUS

California Red-legged Frog Rana draytonii Threatened
There is final critical habitat for this species. Your location overlaps the critical habitat. Species profile: https://ecos.fws.gov/ecp/species/2891

Sierra Nevada Yellow-legged Frog Rana sierrae Endangered
There is final critical habitat for this species. Your location overlaps the critical habitat.
Species profile: https://ecos.fws.gov/ecp/species/9529

Fishes

NAME STATUS

Cui-ui Chasmistes cujus Endangered
No critical habitat has been designated for this species.
Species profile: https://ecos.fws.gov/ecp/species/456

Delta Smelt Hypomesus transpacificus Threatened
There is final critical habitat for this species. Your location is outside the critical habitat.
Species profile: https://ecos.fws.gov/ecp/species/321

Lahontan Cutthroat Trout Oncorhynchus clarkii henshawi Threatened
No critical habitat has been designated for this species.
Species profile: https://ecos.fws.gov/ecp/species/3964

Crustaceans

NAME STATUS

Vernal Pool Fairy Shrimp Branchinecta lynchi Threatened
There is final critical habitat for this species. Your location is outside the critical habitat.
Species profile: https://ecos.fws.gov/ecp/species/498

Vernal Pool Tadpole Shrimp Lepidurus packardi Endangered
There is final critical habitat for this species. Your location is outside the critical habitat.
Species profile: https://ecos.fws.gov/ecp/species/2246

Flowering Plants

NAME STATUS

Layne's Butterweed Senecio layneae Threatened
No critical habitat has been designated for this species.
Species profile: https://ecos.fws.gov/ecp/species/4062

Pine Hill Flannelbush Fremontodendron californicum ssp. decumbens Endangered
No critical habitat has been designated for this species.
Species profile: https://ecos.fws.gov/ecp/species/4818

Stebbins' Morning-glory Calystegia stebbinsii Endangered
No critical habitat has been designated for this species.
Species profile: https://ecos.fws.gov/ecp/species/3991

Critical habitats

There are 2 critical habitats wholly or partially within your project area under this office's jurisdiction.

NAME STATUS

California Red-legged Frog Rana draytonii Final
https://ecos.fws.gov/ecp/species/2891#crithab

Sierra Nevada Yellow-legged Frog Rana sierrae Final
https://ecos.fws.gov/ecp/species/9529#crithab